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Comparing nitrate leaching between contemporary and regenerative dairy pasture management

A thesis presented in partial fulfilment of the requirements for the degree of

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ABSTRACT

New Zealand's dairy industry is crucial to the national economy, yet it faces significant environmental challenges, particularly nitrate (NO_3^-) leaching from intensive pasture-based farming systems. The concentrated nitrogen (N) in urine patches left by grazing cows is the primary contributor to NO_3^- leaching in dairy pastures. While various mitigation strategies have been explored, many have come with limitations. Lower N input regenerative management has emerged as a potential approach to reduce NO_3^- leaching, particularly when combined with diverse pasture. More diverse pasture swards that include deeper-rooted species and herbs already known to reduce leaching have the potential to mitigate NO_3^- -N loss from NZ dairy systems.

The measurement of NO_3^- leaching in free-draining soils is challenging due to the lack of methods that can accurately collect drainage at a scale representing the variability in a pasture paddock. Therefore, the objectives of this research were to 1) assess the effectiveness of trench lysimeters to measure drainage and NO_3^- leaching in a free-draining Manawatū sandy loam soil, 2) compare NO_3^- concentrations measured using trench lysimeters with a suction cup array, and 3) evaluate and compare NO_3^- leaching under three pasture management systems: standard pasture with contemporary management (Std-Con), diverse pasture with regenerative management (Div-Reg), and diverse pasture with contemporary management (Div-Con).

The research was conducted over a period of two years (2023 and 2024) at Dairy One farm at Massey University near Palmerston North, utilising 12 trench lysimeters and 90 suction cups on three different pasture treatments, namely Std-Con, Div-Reg and Div-Con. Contemporary management follows DairyNZ best practices with lower post-grazing residuals and uses mineral/synthetic fertilisers and chemical sprays as needed. Regenerative management involves

longer grazing intervals, higher post-grazing residuals, and reduced use of mineral/synthetic fertilisers and chemical sprays.

The four trench lysimeters per treatment reliably measured drainage depths with generally low standard errors across different drainage events, indicating their reliability for measuring drainage in free-draining soils. Compared to trench lysimeters, suction cups were less effective, recording significantly lower NO_3^- -N concentrations with higher variability. In cases where suction cup NO_3^- -N concentrations were high; this was likely due to individual urine patches resulting in large variability between replicates.

In both years, the Div-Con treatment had the highest NO_3^- -N load, followed by Std-Con and Div-Reg; however, unusually elevated NO_3^- -N concentrations in two lysimeters associated with the Div-Con treatment were responsible for this effect. Therefore, further investigation is required to verify the results from this treatment. In 2023, NO_3^- -N loads were 3 and 14 kg N/ha for Div-Reg and Std-Con, respectively. In 2024, the values increased to 5 and 24 kg N/ha. Both methods indicated a declining trend in NO_3^- -N concentrations across all treatments as the drainage season progressed. Compared to the Std-Con treatment, the Div-Reg treatment consistently measured only 21.4% and 20.8% of NO_3^- -N leaching in 2023 and 2024, respectively, but further monitoring is needed to address the challenges identified in this study and to more thoroughly assess differences between treatments.

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COMMON ABBREVIATIONS

The abbreviations are defined at the first use and then without definition throughout this Thesis.

AMN	Anaerobically mineralisable N
AOA	Ammonia-oxidizing Archaea
AOB	Ammonia-oxidizing bacteria
AWHC	Available Water Holding Capacity
CP	Crude protein
DCD	Dicyandiamide
Div-Con	Diverse Contemporary
Div-Reg	Diverse Regenerative
DM	Dry matter
EC	Electrical conductivity
GHG	Greenhouse gas
HWEC	Hot Water Extractable Carbon
MAL	Maximum acceptable level
ME	Metabolisable energy
MfE	Ministry of Environment
N	Nitrogen
N ₂	Di-nitrogen gas
N ₂ O	Nitrous oxide
NH ₃ -N	Ammonia nitrogen
NH ₃	Ammonia
NH ₄ ⁺	Ammonium
NIWA	National Institute of Water and Atmospheric Research

NO ₂ ⁻ -N	Nitrite nitrogen
NO ₂ ⁻	Nitrite
NO ₃ ⁻ -N	Nitrate nitrogen
NO ₃ ⁻	Nitrate
N ₂ O	Nitrous oxide
NOB	Nitrite-oxidizing bacteria
NZ	New Zealand
P	Phosphorus
PAN	Potentially available nitrogen
PAW	Profile available water
PKE	Palm Kernel Expeller
RA	Regenerative agriculture
RAW	Readily available water
Std-Con	Standard Contemporary
TAW	Total available water
TIN	Total inorganic N
WHC	Water-holding capacity

CHAPTER 1: GENERAL INTRODUCTION

1.1 Background

The dairy industry plays a vital role in New Zealand's (NZ) economy and is the largest contributor to primary industry export earnings. In the year ending March 2024, dairy exports reached \$23.7 billion, accounting for 24% of the nation's total export revenue (Treasury, 2024). NZ's dairy sector predominantly operates on year-round grazing systems (Chapman et al., 2024; Luo & Ledgard, 2021), with dairy cows feeding on perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) pastures (Crush et al., 2005). This system is well-suited to the country's temperate climate, abundant rainfall, and fertile soils (Caradus et al., 2023; Chapman et al., 2024). The increasing global demand for dairy products, economic incentives, and advancements in agricultural technology have driven the intensification of the dairy farming sector, supported by high-input systems that include high-stocking rates, synthetic fertilisers, irrigation, and supplementary feed (Ledgard & Luo, 2021). However, this intensification has also raised environmental concerns, including increased greenhouse gas emissions and the degradation of freshwater quality (Smith & Monaghan, 2020). Therefore, despite its economic significance, NZ's dairy farming sector faces increasing pressure to address water quality issues caused by high nutrient concentrations from dairy farms, which have become a major public criticism, prompting stricter regulations (MfE, 2023b). Elevated NO_3^- concentrations in freshwater can cause eutrophication, resulting in water quality deterioration (Joy et al., 2022). In addition, NO_3^- contamination of drinking water exceeding 11.3 mg N/L poses significant health risks to humans and aquatic life (Ward et al., 2018).

Nitrogen (N) losses from soils to water mainly occur in the form of NO_3^- , through leaching. Urine patches from grazing dairy cows are the primary source of NO_3^- leaching under dairy pastures, accounting for up to 90% of N leached into groundwater (Ledgard et al., 2009). These

urine patches create concentrated areas of N deposition, often exceeding plant uptake capacity, leading to increased NO_3^- mobility and environmental contamination. Dairy cows excrete 70-95 % of the N they consume (Oenema et al., 2005), with the N-loading rates typically ranging from 200 to 1000 kg N/ha (Di & Cameron, 2002b; Selbie et al., 2015). However, pastures cannot uptake these high N inputs from urine and N fertilisers, resulting in elevated NO_3^- concentrations in the soil and increased NO_3^- leaching.

Managing NO_3^- losses to waterways remains a significant challenge in NZ. To address this, various strategies have been implemented to minimise NO_3^- leaching while sustaining dairy farm productivity. These include integrating alternate pastures with species like chicory and plantain (Box et al., 2017; Bryant et al., 2020; Navarrete et al., 2018; Nguyen et al., 2022; Pinxterhuis et al., 2024; Totty et al., 2013), using low-protein supplementary feeds (Edwards et al., 2007; Ledgard et al., 2015; Wilkinson & Waldron, 2017), applying nitrification inhibitors (Di & Cameron, 2016), and implementing duration-controlled grazing (Christensen et al., 2019a). However, some current mitigation practices face several challenges, for instance, the implementation of restricted grazing practices is limited by the cost of suitable standoff facilities and nitrification inhibitors like DCD have been found to result in residues in skim milk powder, leading to manufacturing withdrawals (MPI, 2013; Welten et al., 2016). Moreover, alternative pasture species like chicory and plantain have been found to reduce NO_3^- loss, yet chicory and plantain may accumulate toxic cadmium concentrations (Anderson et al., 2022; Cavanagh et al., 2016; Ubeynarayana et al., 2021), posing animal and human health risks. As a result, a more sustainable solution is needed to manage the economic, environmental, and public health impacts of NO_3^- leaching from the NZ dairy industry.

Regenerative agriculture (RA) has gained recognition as a strategy to tackle environmental challenges by prioritising soil health, carbon sequestration and efficient resource use, which aims to improve and restore ecosystem health, while increasing agricultural productivity (Giller

et al., 2021; Khangura et al., 2023; O'Donoghue et al., 2022). One of the key principles of RA is enhancing pasture diversity through the use of diverse pasture swards containing grasses, legumes, and herbs (Jordon et al., 2022). Diverse pastures can increase root depth and biomass while simultaneously boosting biological N input through the inclusion of legume species (Grelet et al., 2021; Khangura et al., 2023). This is in combination with pasture species that have been proven to reduce NO_3^- leaching, such as plantain and chicory (Bryant et al., 2020), which creates a promising opportunity to address the challenge of NO_3^- loss in NZ's dairy industry.

Measuring NO_3^- leaching under dairy grazed pastures is a complex process affected by various factors, such as animal behaviour, climate conditions, soil properties, interactions between above- and below-ground organisms, and pasture management strategies (Vibart et al., 2016). Studies on NO_3^- leaching in free-draining soils are further challenging due to the rapid movement of water through the soil profile, lower water holding capacity and the limited area that can be instrumented for drainage measurements (Lilburne et al., 2012). Moreover, the spatial and temporal distribution of urine patches in pastures is inherently complex due to the uneven distribution of urine across paddocks due to urine and non-urine patch areas and patches of different ages, volumes and concentrations (Li et al., 2012; Selbie et al., 2015). The measurement of leaching under free-draining soils using trench lysimeters is a relatively new technique, and thorough testing under a grazing system and in comparison, with existing methods such as suction cups is needed.

1.2 Research aims and objectives

The research study objectives were to:

1. To examine trench lysimeters as a method to measure drainage and NO_3^- -N leaching on free-draining Manawatū Sandy loam soils.
2. To compare suction cups and trench lysimeters as two NO_3^- -N leaching measurement methods.
3. Measure and compare NO_3^- -N leaching loads under three different treatments (standard pasture with contemporary management, diverse pasture with regenerative management and diverse pasture with contemporary management).

CHAPTER 2: LITERATURE REVIEW

2.1 Overview of the New Zealand dairy industry

New Zealand (NZ) is the world's eighth-largest milk-producing country (Fernandez-Perez et al., 2022). The dairy industry is NZ's major export industry, exporting 95% of its dairy production to 130 countries worldwide (DCANZ, 2024), making it a significant contributor to the national economy. The average annual export income of the dairy industry in NZ in 2024 was around NZ\$ 23 billion, accounting for 24% of total export values (Treasury, 2024). The NZ dairy farming sector is primarily a pasture-based agriculture system (Luo & Ledgard, 2021), where dairy cattle are mostly fed year-round on perennial pastures (Christensen et al., 2019a; Crush et al., 2005), and on average, 80% of their feed consists of pasture (Ledgard et al., 2020). For the past two decades, this pasture-based dairy farming system has intensified, with higher stocking rates, greater use of nitrogen (N) fertilisers and brought-in feeds, such as imported grains, maize silage, and Palm Kernel Expeller (PKE) (Luo & Ledgard, 2021). Higher stock numbers have increased soil compaction (Laurenson et al., 2017; Pow et al., 2014) but have also increased the amount of dung and urine deposited onto pastures (Moir et al., 2011), while the higher input of N fertilisers has also increased N return to the soil. Furthermore, growing annual fodder crops commonly involves cultivation, which increases organic N mineralisation, leading to a greater risk of nitrate (NO_3^-) leaching (Smith & Monaghan, 2020).

A consequence of the expansion and intensification of the NZ dairy industry has been increasing public concern due to its negative impacts on water quality, notwithstanding the contribution of the sector to the country's economy. As a result, the government introduced an environmental regulatory framework, which slowed the acceleration of dairy farm intensification over recent years. For instance, the national N fertiliser cap is one of the main

regulations that has been introduced through the National Environmental Standard for Freshwater Management, to manage the influence of N loss on the waterways. This includes limiting the use of synthetic N fertiliser (190 N kg/ha/yr) to grazed pastures and requires dairy farmers to report annual usage (MfE, 2023c). A recent report based on Southland and Canterbury shows that total N cycling within farm systems has decreased on average by 17% and 8%, respectively, after implementing the N cap regulation (Journeaux et al., 2023).

2.1.1 Characteristics of typical pasture systems in NZ

Pastoral farming is a key element of NZ's agriculture sector, particularly in dairy, beef and sheep farming. Before the European settlers arrived in the early 1800s, the vast majority of NZ's land area was occupied by rainforests and tussock grasslands (Haggerty & Campbell, 2008). More recently, it has been replaced by perennial pasture and legumes, which currently dominate about 39% of the country's total land area (MfE, 2021). Pasture-based agriculture systems are highly compatible with the temperate climate, abundant rainfall and productive soils of NZ (Caradus et al., 2023). Pastoral farming predominantly comprises perennial ryegrass (*Lolium perenne L.*) and white clover (*Trifolium repens L.*) pastures (Bryant et al., 2010; Davies, 2001), which are favoured for their productivity and grazing tolerance (Cartmill & Donaghy, 2024). These pastures are also preferred for their nutritional value and high metabolisable energy and crude protein (CP) content (Waghorn et al., 2007).

Pasture legumes, like white clover, increase soil fertility by fixing N from the atmosphere and returning some of this N to the soil via plant decomposition and livestock excreta. Some of this soil N can then be accessed by ryegrass, which increases its N content and, therefore, CP in the sward (Davies, 2001). The proportion of white clover in pastures typically ranges from 2-20% of the total dry matter. The lower clover proportions are mainly created by improper

management and environmental stresses during spring and summer (Brock & Hay, 2001; Caradus et al., 1995; Tozer et al., 2014). When pastures are managed properly, they can fix substantial amounts of N, typically ranging from 150 - 250 kg N/ha/year under favourable conditions (Ledgard et al., 2001). However, a review by Brock and Hay (1995) indicates that achieving high overall pasture productivity while maintaining a significant proportion of white clover is difficult, even under optimal management conditions. Moreover, most of this fixed N is returned to the soil as excreta of grazing animals, which is not uniformly applied and results in small areas of high N concentration in the soil (Bowatte et al., 2006). In their study, Chapman et al. (2018a) suggested that increasing the proportion of clover in mixed pastures could elevate N leaching due to the higher N content in clover herbage, which leads to greater N excretion in animal urine.

Standard ryegrass and white clover pasture systems typically contain more than 200 g CP/kg DM and have metabolisable energy (ME) levels greater than 11.5 MJ/kg (Dalley & Geddes, 2012; Dewhurst, 2006; Waghorn, 2007). Dairy pastures often have crude protein levels exceeding 20% of dry matter, corresponding to N concentrations ranging from 3.4-3.9% (Vogeler et al., 2016; Wilkinson & Waldron, 2017). This amount is comparatively higher than other concentrate supplement feeds (1-3%) (Burke et al., 2008). Therefore, this often exceeds the daily dietary protein requirement for lactating cows (Bryant et al., 2020; Wheeler, 2018), especially in spring. It also results in large losses of N in urine at concentrations above plant N requirement, which risks N loss to freshwater, as well as atmospheric loss of N as ammonia (NH₃) and nitrous oxide (N₂O).

2.1.2 Environmental implications of NO₃⁻ leaching from dairy pastures

Currently, the dairy industry poses a substantial challenge to freshwater quality (MfE, 2023b). It is the key source of surface and groundwater nutrient contamination, facilitated by excess nutrient loss from farms, particularly N and phosphorus (P) (McDowell et al., 2017). Elevated NO₃⁻ concentrations can stimulate unwanted microbial and algae growth in surface waters, such as rivers and lakes (Joy et al., 2022), causing eutrophication. Some of the types of algae (blue-green algae or cyanobacteria) are toxic to humans and animals, and the nuisance growth of these plants and algae also causes turbidity and sedimentation, reducing the recreational and cultural values of surface water resources. Freshwater systems are highly sensitive to N and P; therefore, even minor increases can create noticeable growth of undesirable aquatic organisms that cause detrimental effects (Cameron et al., 2013).

Excess nutrients, primarily NO₃⁻ leaching from dairy farms, have also impacted groundwater quality targets in NZ. The NO₃⁻ contamination of drinking water brings serious health risks, such as methemoglobinemia (blue baby syndrome) in bottle-fed babies (Ward et al., 2018). High NO₃⁻-N concentrations in drinking water are also associated with an increased risk of colorectal cancer (Espejo-Herrera et al., 2016; Schullehner et al., 2018), which is one of the most common types of cancer in NZ (CCA, 2024). Therefore, there is concern among the public about increasing NO₃⁻-N concentrations in groundwater that exceed the standard N concentration for drinking water. The NZ government has introduced a Maximum Acceptable Level (MAL) guideline for NO₃⁻ in drinking water of 11.3 mg/L NO₃⁻-N, in line with the recommendation from the World Health Organisation (MfE, 2023a).

Moreover, regional authorities routinely examine NO₃⁻-N concentrations in groundwater and freshwater sources. This includes providing trend assessments, where (trends are classified as improving, worsening or indeterminate) the latter being used when no trend direction or not enough statistical certainty is present to determine trend direction. A recent NZ trend

assessment of 20 years (2001–2020) of river water quality for NO_3^- -N showed that 38% of monitored sites were improving, 54% were worsening, and 8% were indeterminate (Figure 2.1) (Stats, 2022).

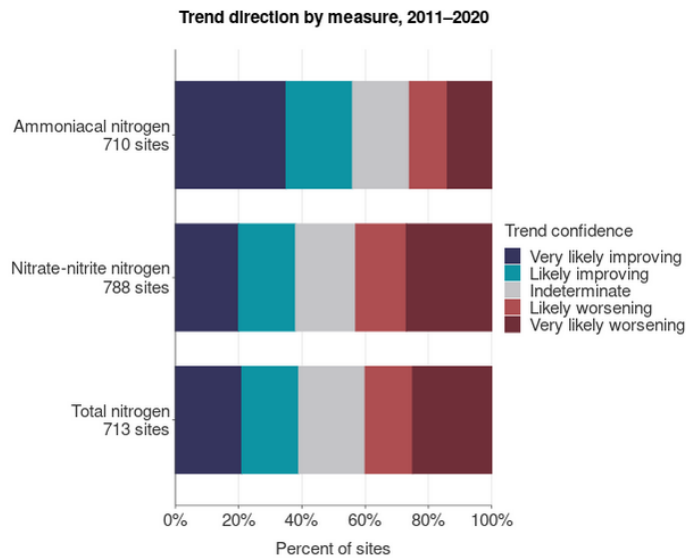


Figure 2.1: River water quality trend direction measure, 2011-2020 (Stats, 2022)

2.2 Overview of sources and processes of NO_3^- loss from dairy pasture soils

Nitrogen is one of the main nutrients to determine plant growth (Caradus et al., 2023), along with P and potassium (K), which can be lost from soils to water primarily in the form of NO_3^- via leaching. NO_3^- is an anion, a readily available form of N for plant uptake (Caradus et al., 2023), that easily moves through NZ’s negatively charged soils with drainage water due to their higher mobility and water solubility (Ledgard et al., 2000; Wang & Li, 2019) reaching groundwater and then surface water bodies (Di & Cameron, 2018). There are four major pathways by which N can be added to a dairy farm soil: livestock excreta (urine and dung), fertiliser N, effluent applications and the N fixation by legumes (Bolan et al., 2004). Concentrated urine patches caused by intensively grazed dairy cows have been identified as the main ‘hot spots’ of NO_3^- leaching in dairy-grazed pastoral systems (Selbie et al., 2015), which are responsible for up to 90% of N leached to groundwater (Ledgard et al., 2009).

Furthermore, inappropriate timing and rate of synthetic N fertilisers and effluent can lead to poor utilisation of N by pasture (Bryant et al., 2020; Cameron et al., 2013), creating elevated NO_3^- concentrations in the soil. This includes excessive N fertiliser application rates or applications prior to heavy rainfall, which can lead to greater NO_3^- losses. According to recent statistics, the dairy sector applies 62% of NZ's total N fertilisers, with the balance applied by other agricultural industries (Fertiliser Association of New Zealand, 2022).

Soil surplus N can be lost via drainage under high rainfall and low evapotranspiration (Cameron et al., 2013; Selbie et al., 2015), particularly in late autumn, winter and spring (Cameron et al., 2013). A study conducted in the Waikato region of NZ, on a well-drained Otorohanga soil (Vogeler et al., 2014) using the Agricultural Production Systems Simulator (APSIM), found that at an annual application rate of 220 kg N/ha, direct leaching of fertiliser N surpassed that from urine patches in only one out of ten years. Therefore, even when high annual application rates of N fertiliser are used, the loss of N in leaching will still be greater from urine patches in most years. (Fertiliser Association of New Zealand, 2022). This confirms that the direct impact of N fertilisers on leaching from pasture is typically minimal and can largely be reduced by avoiding applying N fertiliser during winter (Vogeler et al., 2016).

2.3 N cycle in pastoral soils

The N cycle is vital to soil health and productivity, particularly in pasture-based agricultural systems. Over 95% of soil N is present in the plant-unavailable organic forms (Cameron et al., 2002; McLaren & Cameron, 1996), which needs to be broken down into inorganic forms (NH_4^+ and NO_3^-) for plants to uptake (Bolan et al., 2004). The soil N cycle (Figure 2.2) consists of several transformations such as N fixation, mineralisation, immobilisation, nitrification, denitrification, NO_3^- leaching, and ammonia (NH_3) volatilisation and produces different forms of N that can either be plant available, immobilised in the soil organic fraction or lost from the system in leachate or as gas.

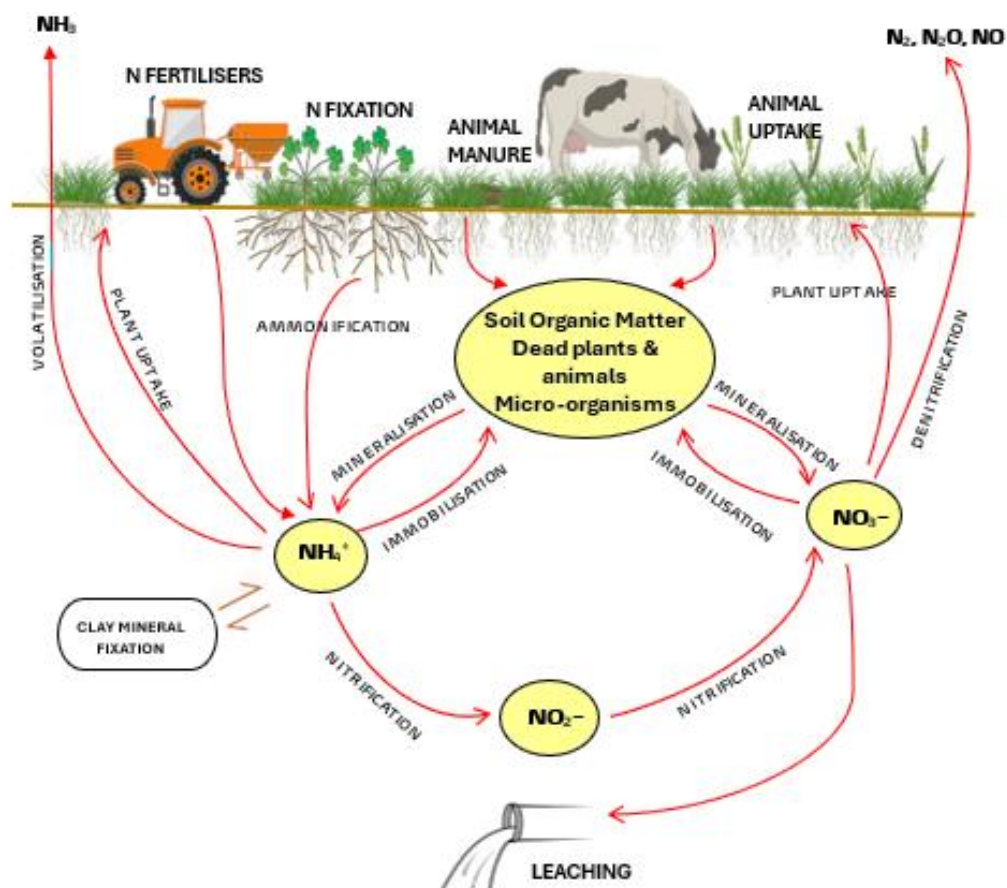


Figure 2.2: Nitrogen cycle in pastoral soil - Based on Di and Cameron (2002b)

2.3.1 Nitrogen fixation

Nitrogen fixation mostly occurs during the main pasture growing seasons, particularly in the spring and autumn (Caradus et al., 1996). It influences the overall soil N budget by providing a natural source of N for plant growth in NZ dairy farms. In ryegrass and white clover pastures, N fixation primarily occurs through white clover, which forms symbiotic relationships with N-fixing bacteria (*Rhizobium* species) (Bolan et al., 2004; McLaren & Cameron, 1996). These bacteria convert atmospheric N into NH_4^+ while living in root nodules of legumes in exchange for carbohydrates produced by the plant (McLaren & Cameron, 1996). Some free-living bacteria in the rhizosphere, such as *Azotobacter*, have also been found to fix N without a true symbiotic relationship with the plant (McLaren & Cameron, 1996).

The amount of N fixed depends on several factors such as legume type, the proportion in the pasture sward, soil and climatic conditions, available nutrients, and grazing management (McLaren & Cameron, 1996). For instance, N fixation slows during winter when cold temperatures and low microbial activity reduce the symbiotic process (Kantar et al., 2010). Moreover, biological N fixation often decreases when soil N levels are high (Ledgard et al., 1996; Walker, 1995) because the plant relies more on available N in the soil rather than its symbiotic relationship with rhizobia. Numerous studies which have investigated the interaction of N fertiliser and legumes (white clover) and its influence on N fixation in dairy-grazed pastures in NZ, showed the application of N fertiliser generally results in a decline in the proportion of legumes in a sward (Chapman et al., 2018a; Lambert et al., 2012). According to Ledgard et al. (2001), N fertiliser reduced the average proportion of fixed clover N from 77% in the untreated pasture to 43–48% in the pastures receiving 400 kg N/ha. Subsequently, the total N fixed declined from 154 kg N/ha to a range of 39–53 kg N/ha annually.

2.3.1 Mineralisation/ immobilisation

Mineralisation refers to the conversion of soil organic N, which is derived from animal excreta, plant residuals and other organic materials, into inorganic N (Cabello et al., 2009; McLaren & Cameron, 1996). N mineralisation from animal urine is usually rapid because most of the urine is in the form of urea. This urea N mostly converts into NH_4^+ via a process called ammonification within about 24 hours after urine deposition onto pasture (Bolan et al., 2004; McLaren & Cameron, 1996). Mineralisation rates of soil organic N can be influenced by soil temperature, moisture, and oxygen availability. For instance, mineralisation is constrained during the cool winter period and increases when soil temperature is higher in the spring (Beare et al., 2022). Moreover, the process of mineralisation continues despite pasture growth slowing or pastures becoming dormant due to soil moisture deficit in summer and autumn, which leads to a significant accumulation over these seasons (Powlson, 1993). In contrast, immobilisation occurs when microorganisms ingest inorganic N from the soil to meet their metabolic needs, converting it back into plant-unavailable organic forms (Bolan et al., 2004; McLaren & Cameron, 1996).

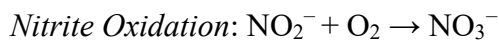
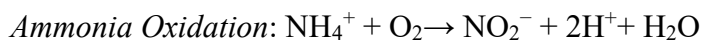
2.3.2 Volatilisation/urea hydrolysis

Volatilisation is the loss of N from soil as ammonia gas (NH_3), typically from applied urea fertilisers, effluent applications, or urine patches in pastoral systems (Smith et al., 2020). Urea hydrolysis occurs as the initial breakdown of urea into ammonium (NH_4^+), which plants or microbes can utilise, leading to a temporary increase in soil pH, which in turn can improve NH_3 volatilisation (Haynes & Williams, 1993; McLaren & Cameron, 1996). Therefore, volatilisation can be greatest within the first few days after urea fertiliser or urine application. This process is affected by several factors, including soil moisture, temperature, and soil pH.

For instance, drier soil conditions, typically in summer, accelerate the conversion of urea to NH_4^+ , further elevating soil pH, thereby increasing the risk of NH_3 volatilisation (Bishop & Manning, 2010).

2.3.3 Nitrification

Nitrification is a two-step process that first converts NH_4^+ to NO_2^- involving ammonia-oxidizing bacteria (AOB) and ammonia-oxidizing Archaea (AOA), such as *Nitrosomonas* species and then, and then converts NO_2^- to NO_3^- by nitrite-oxidizing bacteria (NOB), such as *Nitrobacter* (Cameron et al., 2013; Gwak et al., 2019; Lourenco et al., 2022).

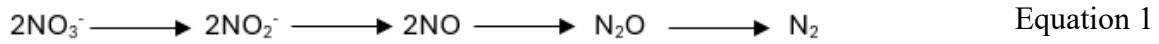


Unlike NO_3^- , NH_4^+ is relatively immobile in soil. Nitrate via nitrification is plant-available and significantly increases the risk of NO_3^- leaching, particularly if NO_3^- accumulation in the soil coincides with a period of soil drainage. Nitrification rate can be dependent on soil moisture, pH, temperature, wind velocity and the nutrient status of the soil (McLaren & Cameron, 1996). For instance, in soils that are very wet or very dry, the nitrification rates can be lower (Cameron et al., 2013). Conversion of NH_4^+ to NO_3^- in urine patches or from urea fertiliser generally occurs within 7 to 14 days (Otene et al., 2024).

2.3.4 Denitrification

Denitrification refers to an anaerobic process involving the microbial conversion of NO_3^- to di-nitrogen gas (N_2) or nitrous oxide (N_2O) released into the atmosphere (Saggar et al., 2013). It is the opposite of the nitrification process and a key part of the pastoral N cycle. It plays a

valuable role in agriculture and the environment, removing excess N from soil and water systems. However, if the denitrification process is incomplete, the outcome can be N₂O (Eqn. 1), which is a potent greenhouse gas.



The soil's denitrification rate depends on the availability of labile C, temperature, pH and redox potential of soil microsites (Bolan et al., 2004; McLaren & Cameron, 1996; Saggar et al., 2013). Denitrification is usually slower at low soil pH levels and higher at warmer soil temperatures or under waterlogged conditions (McLaren & Cameron, 1996). Moreover, the denitrification rate is higher in grazed pasture lands compared to ungrazed pasture lands (Luo et al., 1999) due to the deposition of dung and urine of grazing animals. This is because the main source of denitrification in grazed pastures is the livestock urine patch due to the high concentrations of urea that mostly converts to NO₃⁻ over time. This NO₃⁻ is at risk of denitrification, especially when the soil becomes saturated, which commonly occurs during winter and early spring.

2.3.5 Plant uptake

The N uptake of pastures from the soil mainly occurs in the form of NO₃⁻ or NH₄⁺ (inorganic N) (Bolan et al., 2004), which is absorbed from the soil solution through the root to the xylem. The NO₃⁻ absorbed by plants may be transferred within the plant unchanged or converted to NH₃ within the root. This NH₃ is then transformed into amino acids, amines, or amides to be translocated through the plant to produce organic compounds such as proteins and nucleic acids (McLaren & Cameron, 1996). Pasture uptake of N can be high compared to other crops (Whitehead, 1995). On average, a well-managed ryegrass-white clover pasture typically takes up between 200 to 400 kg N/ha/yr (Di et al., 1998), depending on factors such as climate, land type, soil fertility, and management practices. Pasture N uptake tends to be higher during

warmer seasons, particularly in spring, when pasture growth is more rapid (Whitehead, 1995), and longer sunlight hours support increased uptake of N to enhance photosynthesis (Selbie et al., 2015). If unlimited N is available in the soil, pasture may also take up N above requirements (exceeding a plant tissue concentration of 4%), termed 'luxury uptake' (Selbie, 2014).

2.4 Factors influencing NO₃⁻ leaching from dairy-grazed pasture

Nitrate leaching refers to the transport of NO₃⁻ down through the soil profile and below the root zone due to excess drainage from rain or irrigation. The NO₃⁻ concentration in soil and the water flux are the main factors that influence leaching under grazed pastures (Di & Cameron, 2002b; Selbie, 2014). In NZ, NO₃⁻ leaching mainly occurs during late autumn, winter and early spring, when there is a surplus of rainfall over evapotranspiration and the soil moisture is at field capacity (Christensen et al., 2019b; McLaren & Cameron, 1996). This is also the period during which plant N uptake is generally low. Several studies (Cameron et al., 2013; Di & Cameron, 2002b; Jarvis, 2000) show that NO₃⁻ leaching from grazed pastures can also be influenced by soil texture and drainage characteristics, climate, season and drainage, and especially time and the rate of N inputs. Due to drainage, this excess N can be lost via leaching, primarily in the form of NO₃⁻, which accounts for up to 99% of total mineral N leached (Shepherd et al., 2011).

2.4.1 Role of the cow urine patch

Cow urine patches are the single largest contributors to NO₃⁻ leaching under dairy pastures (Di & Cameron, 2002a; Di & Cameron, 2007). The N concentration in cow urine is higher (3 to 20.5 g N/kg) compared to dung (3.4 to 5.0 g N/kg) (Dijkstra et al., 2013), and over 70% of the N excreted is via urine (Zaman et al., 2009). Most of this urine (typically about 70-80%) is

urea-N (Dijkstra et al., 2013; Zaman et al., 2007) and the rest is made up of amino acids and peptides, which are readily mineralisable and readily leached to groundwater (Haynes & Williams, 1993; Ramírez, 2017).

Grazing animals usually urinate up to 8-12 times per day on average (Di & Cameron, 2002b; Haynes & Williams, 1993; Moir et al., 2011). N-loading rates can more typically range from 200 to 1000 kg N/ha, with a mean of 613 kg N/ha (Di & Cameron, 2002b; Selbie et al., 2015). They are often visible as darker green patches in the paddocks and show increased pasture growth (Moir et al., 2011). According to the study findings of Moir et al. (2011), the surface area of urine patches ranged from 0.03 to 1.1m², with the mean surface area of the urine patch ranging from 0.34 to 0.40 m² (Haynes & Williams, 1993; Moir et al., 2011). The volume of urination is highly variable in cows, varying from 0.30 to 7.83 L/event (Betteridge et al., 2013) with an average of about 2.1 L/event (Selbie et al., 2015). Over a year, urine patches can cover about 20-30% of a paddock's area (Moir et al., 2010). Moreover, the N concentration of cow urine is influenced by their reproductive status and time of day (Bryant et al., 2014; Hoogendoorn et al., 2010).

2.4.2 Seasonal variation

A previous study (Moir et al., 2011) noted slightly smaller urine patches in winter and autumn compared to spring or summer. The seasonal variation of urine patch sizes can be due to the seasonal variation of the crude protein content of animal feed, water intake and cow number (Litherland et al., 2002; Moir et al., 2011) The main drainage period in NZ is from May to October, coinciding with low transpiration rates and many rain events (Di & Cameron, 2002a; Ledgard et al., 2009; Shepherd & Lucci, 2013; Silva et al., 1999), which enhances the risk of NO₃⁻ loss.

Grazing events during late summer and autumn periods with limited rainfall and no irrigation may especially allow significant accumulation of urinary N in surface and subsurface soils, and these contribute to a carry-over of soil NO_3^- , which can be leached the following winter (Di and Cameron., 2002a). However, irrigation during summer and autumn, maintains pasture growth and enhances pasture N uptake, which reduces the accumulation of N from urine patches before the onset of winter drainage. For instance, in a study conducted under flood-irrigated dairy pasture, with free-draining soil in the mid-Canterbury area of NZ, Di and Cameron (2002a) showed that the NO_3^- leaching from urine patches deposited in autumn is higher (38-58%) than those applied in spring (29%). Another study (Shepherd et al., 2011) conducted on free-draining soils in the Waikato region found that between 40% and 50% of the cow urine N deposited on pasture from February to May was leached by the end of the winter. Furthermore, the cow urine deposited at the end of spring remained in the soil until the following autumn, causing elevated N to be stored in the soil profile (Shepherd et al., 2011). A recent study by Beale et al. (2023) observing the seasonal effect of cattle urine deposition on stony silt loam soils in Canterbury also showed late summer and early autumn (February–March) urine deposition led to higher NO_3^- leaching, driven by rapid nitrification and limited plant N uptake.

Therefore, any fertiliser N applications are recommended to coincide with periods when the pasture is actively growing to maximise pasture N uptake and DM yield and reduce the surplus of soil NO_3^- which is at risk of leaching (Ledgard et al., 2009). Both pasture N content and the soil moisture content generally reach maximum in spring, leading to higher N concentrations in soil due to increased rainfall and rising soil temperatures, which also promote mineralisation. However, the lowest risk of leaching occurs in the spring because the optimal climate conditions during this period encourage rapid pasture growth, driving uptake of N by pastures (Anderson et al., 1997; Christensen et al., 2019a; Shepherd & Lucci, 2013). In addition,

increased denitrification in warm wet soils can also lead to lower soil NO_3^- -N concentrations in spring (Luo et al., 2008; Saggar et al., 2004).

2.4.3 Cow dietary N intake

Several studies have shown that the N concentration in pastures highly depends on the season, fertiliser N application rate and period between application and grazing (Cameron et al., 2013; Christensen et al., 2019a; Selbie et al., 2015; Shepherd & Lucci, 2013). Also, urinary N concentrations are strongly linked with cow N intake (Castillo et al., 2000). Therefore, the amount of N returned in urine can increase when pasture N concentrations are elevated after N fertiliser application (Hoekstra et al., 2008). Pasture tends to take up a surplus of N in the first few weeks following N fertiliser application, leading to accumulated N in pasture swards (Jacobs et al., 2002). However, some studies (Shepherd & Lucci, 2013; Thomas et al., 2016) suggest that N fertiliser has a small effect on pasture N concentrations.

Furthermore, pasture N concentrations can be influenced by clover composition and usually increase with increasing clover content due to higher crude protein concentration in clover (Ledgard, 2001). Dairy pastures in NZ typically have higher (4%) N concentrations compared to concentrate supplements (1-3%) due to the clover content in pastures (Burke et al., 2008). Consumption of this pasture, which often exceeds the dietary crude protein requirement of dairy cattle, especially in spring, results in excreting a large portion of N in their urine. Furthermore, supplementary feeds with high N content, such as grass silage, increase the N intake of cows, leading to higher N excretion in urine. This elevated N excretion contributes to greater NO_3^- leaching from urine patches. Substitution of high protein supplementary feed by lower N feeds (i.e. maize silage, fodder beet) can significantly reduce NO_3^- leaching losses (Beukes et al., 2024).

2.4.4 Soil factors

The spatial variability of soils with different soil drainage classes is one of the key factors determining soil water drainage and, thus, NO_3^- leaching (Burkitt, 2014). This includes changes in both the physical and chemical properties of soils. Fine-textured soils (silt and clay soils) have poorer drainage potential and, thus, lower risk of NO_3^- leaching than coarse-textured (sandy and stony) well-drained soils. Due to large pore spaces and greater hydraulic conductivity (the rate at which water moves through soil), sandy soils have a higher propensity for NO_3^- leaching than clay or silt loam soils (Cameron et al., 2002). NO_3^- leaching via drainage in free-draining soils is caused by heavy rainfall and over-irrigation (Selbie et al., 2015), particularly during late autumn and winter when evapotranspiration is low and soil moisture content is high (Cameron et al., 2002). Moreover, due to the rapid draining in free-draining soils, there is less likelihood of plant uptake, denitrification, and immobilisation, which remove NO_3^- from pasture soils, thus increasing NO_3^- leaching losses (Cameron et al., 2002; Carrick et al., 2013a). For instance, a previous study by Di and Cameron (2002a) found that NO_3^- leaching under free-draining shallow stony soils (e.g. flood plains of Canterbury) increased with irrigation.

Profile available water (PAW) or water-holding capacity (WHC) is a specific soil moisture property that significantly impacts both NO_3^- levels in the soil and the drainage within the root zone (Cichota et al., 2013). Together, higher inorganic N and lower PAW lead to greater NO_3^- leaching. The NO_3^- leaching can also be influenced by soil profile depth, compaction and the quality and quantity of organic matter (Di & Cameron, 2002a; McLaren & Cameron, 1996). In particular, Lilburne and Webb (2002) showed that the leaching risk is higher in shallow and stony soils than in moderately deep to deep and non-stony soils, specifically in arable systems.

2.4.5 Climate/ irrigation factors

Rainfall and irrigation are crucial factors in determining NO_3^- leaching on dairy farms in NZ. Heavy rain following fertilisation may lead to peaks in NO_3^- loss (Ledgard et al., 1998; Sprosen & Ledgard, 2020). Higher rainfall during the winter to early spring along with low evapotranspiration and lower plant NO_3^- uptake, exacerbates drainage through the soil (Burkitt, 2014). Moreover, when the soils are at field capacity, intense rainfall such as summer storms or surplus irrigation (i.e. flood irrigation) can also trigger leaching at other times of the year (Cameron et al., 2002). This creates a likelihood of greater N loss than those during the typical winter drainage season (Graham et al., 2022; Hoogendoorn et al., 2011; Scholefield et al., 1993; Vogeler et al., 2010). Furthermore, soils under saturated conditions tend to lose NO_3^- via surface runoff, under rainfed or irrigation conditions. Dairy farms with compacted soils may have a higher risk of surface runoff following excessive irrigation (Burkitt, 2014).

In any year when rainfall does not exceed evaporation, NO_3^- leaching can be significantly affected due to limited soil water availability, particularly in dairy farming areas (Cameron et al., 2002). Low rainfall events may result in little or no drainage, which restricts the downward movement of NO_3^- through the soil profile. This phenomenon can also be used in effective irrigation scheduling. Adjusting irrigation practices during dry summer months to give adequate soil moisture to increase pasture growth and N uptake, which is otherwise limited, may decrease NO_3^- leaching losses (Di & Cameron, 2002a; Graham et al., 2022). However, increased pasture growth and N uptake will also increase cow N intake and N returned in urine, which can also increase N leaching risk (Moir et al., 2016). Several studies recommend maintaining soil moisture deficits in irrigation to increase the capacity for soil to hold any rainwater, subsequently increasing water use efficiency and also reducing the risk of NO_3^- leaching (Graham et al., 2022; Thomas et al., 2016).

Similarly, a modelling study using Overseer (Bright et al., 2018) using 12 dairy farms in the Canterbury region of NZ, has shown the possibility of gaining an average of 27% reduction in N loss (~19 kg N/ha/year) by improving irrigation practices, such as changing the irrigation trigger and/or target levels to use irrigation water and rainfall efficiently. This includes adjusting PAW to maintain pasture production while reducing N loss to water through improved irrigation efficiency and effective use of rainfall.

2.5 Opportunities to reduce N leaching from urine patches

Research to date has developed different strategies to reduce NO_3^- leaching in dairy pastures.

2.5.1 Grazing management

Restricted grazing strategies, such as zero/nil grazing and duration-controlled grazing, have proven to be effective in preventing pugging and compaction, nutrient loss and pasture damage (Christensen et al., 2019a). Duration-controlled grazing strategy limits grazing cows by using standoff facilities or animal shelters (Christensen et al., 2019a; de Klein & Ledgard, 2001). The effluent is collected in this method to reduce the amount of N deposited on the pasture and spread at a time that matches plant demand and/or has less risk of drainage and NO_3^- leaching. Zero/nil grazing is known as ‘cut-and-carry’ or ‘stall feeding’, where animals are in sheds all year round. According to de Klein and Ledgard (2001), leaching losses from nil grazing can significantly reduce NO_3^- leaching by 55-65%, and restricted grazing systems can reduce NO_3^- leaching by 35-50%, compared to traditional grazing methods. Similarly, Christensen et al. (2019a) reported that controlled grazing led to a 50% average reduction in NO_3^- leaching. However, implementing restricted grazing practices may require farmers to invest in suitable standoff facilities, which can be cost-prohibitive for some farmers.

2.5.2 Supplementing diet with low-protein forage

The N intake directly influences the N concentration of cow urine (Castillo et al., 2000; Kebreab et al., 2002), particularly rumen-degradable protein (RDP), which can result in elevated N excretion in urine (Pacheco et al., 2024). Feeding cows with low protein dietary supplements has proven to cause a significant decrease in total N in urine and, therefore lower urine N excretion (Misselbrook et al., 2005). For instance, feeding cows with lower protein maize silage (7-8% CP) as a supplementary feed could reduce the urinary N, by lowering the overall protein level in the cow's diet (Wilkinson & Waldron, 2017) and balancing the protein: energy requirement ratio (Gregorini et al., 2010). This involves regulating the inefficiency of N cycling in grazed pastures due to excess protein content compared to the dietary requirement of cows and high N-concentrated urine patches (Ledgard et al., 2000). Additionally, a study (Ledgard et al., 2015) showed that salt supplementation in cow diets could decrease per-hectare N leaching by 10-22% by increasing cattle water intake and urination frequency, thus lowering urine N deposition rate. Moreover, adding high-sugar grass species, such as high-sugar perennial ryegrass, into the cow diet has also been investigated as a mitigation option of NO_3^- leaching (Edwards et al., 2007).

2.5.3 Alternative pasture/forage species

Several alternative pasture species, such as Plantain (*Plantago lanceolata*), Chicory (*Cichorium intybus*) and Italian ryegrass (*Lolium multiflorum*), have been examined in NZ, focusing on strategies to mitigate NO_3^- leaching from dairy farms. These species were studied for their ability to improve N utilisation by cows and/or reduce N excretion, which ultimately lowers N leaching to the environment. Plantain is known as a promising alternative pasture species, which has been shown to reduce NO_3^- leaching by 20-60% when incorporated with

other pastures (at least 30% plantain) leading to lower per-patch N concentrations, diluting cow urine (Box et al., 2017; Bryant et al., 2020; Navarrete et al., 2018; Nguyen et al., 2022; Pinxterhuis et al., 2024; Totty et al., 2013) via increased urine volumes (Marshall et al., 2021; Nguyen et al., 2022). Plantain has low DM content and higher plant water content (Box et al., 2017; Marshall et al., 2021; Navarrete et al., 2018; Nguyen et al., 2022) and also produces secondary compounds, such as aucubin, which can inhibit nitrifying bacteria in the soil and, thus, can be considered a natural nitrification inhibitor (Carlton et al., 2019; Gardiner et al., 2020; Judson et al., 2018). Chicory is similarly effective at reducing urinary N concentration due to higher plant water content and lower DM. Integrating chicory into pastures can reduce N losses from urine patches due to its ability to alter N cycling in the soil and reduce the N concentration in cow urine (Cheng et al., 2015; Mangwe & Bryant, 2022). Moreover, its deeper tap root system absorbs nutrients from deeper soil layers, which can help capture excess N before leaching into groundwater. However, chicory and plantain may accumulate toxic cadmium (Cd) concentrations (Anderson et al., 2022; Cavanagh et al., 2016; Ubeynarayana et al., 2021) from pasture soils, allowing Cd to enter the food chain and pose animal and human health risks. Italian ryegrass can reduce NO₃⁻ leaching primarily due to its enhanced N uptake during cooler seasons (Moir et al., 2013; Woods et al., 2017). According to Maxwell et al. (2019), Italian ryegrass reduced NO₃⁻ leaching by 33-46% compared to other ryegrass varieties. Al-Marashdeh et al. (2021) demonstrated that integrating Italian ryegrass with plantain (*Plantago lanceolata*) into pasture swards further reduced NO₃⁻ leaching, with reductions of up to 56% while maintaining pasture productivity in a two-year study conducted in Canterbury.

2.5.4 Regenerative management and diverse pastures

Regenerative management

Regenerative agriculture management is an approach to farming that aims to improve and restore ecosystem health while increasing agricultural productivity (O'Donoghue et al., 2022). The core principles of regenerative agriculture are promoting longer-standing pastures during grazing, minimising chemical inputs, reducing soil disturbance, optimising photosynthesis efficiency, and fostering pasture diversity (Giller et al., 2021). Enhancing pasture diversity through the introduction of a variety of grasses, legumes, and herbs, combined with strategically implemented extended deferred grazing, promotes improved soil health. The integration of diverse pasture swards as part of regenerative management (Totty et al., 2013) presents an opportunity to reduce N loss from NZ's dairy farming industry. N loss could potentially be decreased through the incorporation of pasture species in the sward that are known to reduce NO_3^- leaching and the incorporation of legumes to capture atmospheric N and reduce reliance on synthetic N fertilisers (Graham et al., 2024; Grelet et al., 2021)

In NZ, regenerative agriculture employs several key principles to reduce NO_3^- leaching (Table 2.1). However, implementing regenerative principles brings unique challenges to the dairy sector, such as initial transition costs, potential short-term yield loss and uncertainty, knowledge-intensive management and monitoring, the persistence of diverse species and weed control and nutrient management issues with organic amendments (Grelet et al., 2021).

Table 2.1: Regenerative agriculture principles implemented in NZ dairy farming based on (Grelet et al., 2021)

Principle	Approach	Description
Reduced synthetic N use	Low N fertilisation	Minimise the need for synthetic fertilisers by enhancing soil fertility with organic matter
	Legume integration in pastures	N-fixing plants supply natural N to the soil, reducing the need for synthetic N
Soil health and organic matter	Compost and organic amendments	Build soil organic matter to improve N retention and reduce leaching
	Permanent pastures/cover crops and diverse swards	Retain soil covered year-round, reducing erosion and NO ₃ ⁻ loss
Efficient grazing management	Rotational grazing	Manage grazing to prevent overgrazing and soil exposure and reduce leaching risk, particularly in winter.
	Grazing exclusion near waterways	Decrease direct nutrient runoff to improve water quality.
Riparian buffer zones	Buffer strips along waterways	Introduce vegetation along waterways to trap nitrates
Efficient water management	Proper use of irrigation	Prevent excess water application to reduce NO ₃ ⁻ leaching

Diverse pastures

Diverse pasture swards typically contain a mix of grasses, legumes and herbs (Figure 2.3) and present an opportunity to potentially decrease NO₃⁻ leaching, especially if this mix includes species known to reduce NO₃⁻ leaching, such as plantain, chicory and Italian ryegrass. Grass species such as tall fescue, cocksfoot, phalaris and Italian ryegrass, legumes like red clover, alfalfa, birdsfoot trefoil, and sulla, and herbs like chicory and plantain are recommended as alternative pasture species for NZ (Chapman et al., 2008).



(a) Perennial ryegrass



(b) White clover



(c) Red Clover



(d) Chicory



(e) Plantain



(f) Lucerne



(g) Prairie grass



(h) Tall Fescue



(i) Cocksfoot

Figure 2.3: Examples of common diverse pasture species (a) Perennial ryegrass (b) White clover (c) Red clover (d) Chicory (e) Plantain (f) Lucerne (g) Prairie grass (h) Tall Fescue (Dairy NZ, 2024) (i) Cocksfoot. All photos with the exception of Tall Fescue have been taken by the author.

Diverse pastures are selected for their favourable functional characteristics to enhance productivity, resilience, and environmental sustainability in dairy systems (Graham et al., 2024). For instance, grasses absorb N efficiently from the upper soil layers, while deep-rooted species, such as chicory and lucerne and some strains of ryegrass, are chosen due to their higher belowground biomass which can potentially enhance soil carbon stocks (McNally et al., 2015)

and the uptake of NO_3^- from deeper soil layers (Beukes et al., 2014; Crush et al., 2005; Crush et al., 2007). Perennial ryegrass–white clover mixtures (standard pastures) typically have shallow root systems, with ~ 80% of the root mass generally growing in the top 15 cm of soil (Haynes & Williams, 1993). Moreover, cold-resistant species are used to increase N uptake during the winter drainage season (Malcolm et al., 2014), and legumes are included to enhance N fixation, thus reducing the use of synthetic fertilisers. However, if there are seasonal or climatic mismatches, such as N fixation occurring when plant demand is low, it can increase the risk of NO_3^- leaching (Peoples et al., 2009). Furthermore, drought-tolerant species such as tall fescue, paspalum, phalaris, and cocksfoot are grown for increased summer growth (Fulkerson & Donaghy, 2001).

However, research in NZ presents mixed findings regarding the impact of diverse pasture swards on NO_3^- leaching compared to standard pastures. For instance, some studies indicate increasing sward diversity can reduce N losses through the urine of lactating cows (Beukes et al., 2014; Edwards et al., 2015; Totty et al., 2013; Woodward et al., 2012) while some studies found no difference in NO_3^- leaching between standard and diverse pastures (Graham et al., 2024). An indoor cattle feeding study by Woodward et al. (2012) compared the feeding of standard pasture and diverse pasture containing chicory, plantain and lucerne and noted a reduction in urinary N concentration in cows fed the diverse pasture by up to 50%. Several studies have compared dry matter production between diverse and standard pastures (Graham et al., 2024; Totty et al., 2013; Woodward et al., 2013) but found no difference or an insignificant reduction of DM in diverse pastures (Beukes et al., 2014; Khaembah et al., 2014). However, the research conducted by Nobilly et al. (2013) noted an increase of 1.62 t DM/ha in diverse pastures than standard pastures, with greater DM production in the summer. Woodward et al. (2013) in the Waikato region found that while the annual DM production was comparable between standard and mixed pasture systems, mixed pastures yielded more DM (12% in first

year and 47% in second year), driven by the enhanced growth of species such as chicory and lucerne in warm, dry conditions. However, this advantage did not extend into the winter months. Hence, generally, grazing standard pastures could lead to higher cow N intake compared to diverse pastures, resulting in higher N return in urine. In another study, McNally et al. (2015) noted that diverse pastures (chicory, plantain, and alfalfa with perennial ryegrass and white clover) had larger and deeper root masses than standard pastures, particularly at greater soil depths, suggesting a potential to absorb more NO_3^- and presumably water from deeper soil layers.

2.6 Measuring NO_3^- leaching in free-draining soils

2.6.1 Drainage variability

The collection and measurement of drainage in the field is expensive and complicated since drainage occurs mostly invisibly underground. Drainage is also variable due to the irregularity of rainfall distribution and the uneven input of nutrients, such as animal urine and dung (Moir et al., 2011). It is also challenging to trap drainage from a whole paddock; therefore, the nutrient loss associated with drainage is generally measured and calculated from small areas and then used to estimate losses per paddock or hectare. Measuring drainage volumes is even more difficult in free-drainage soils than in poorly drained soils (Webster et al., 1993). In contrast to free-draining soils (i.e. Canterbury region NZ), water movement through the soil profile is slower in poorly drained soils with high water-holding capacity, therefore, these soils are well suited to subsurface drainage (i.e. mole and tile drainage), which makes it easier to capture drainage water (Burkitt, 2014). However, due to the rapid drainage in free-draining soils (e.g. stony soils, sands, gravels and pumice soils), which are dominant in many dairy farming areas in NZ, these soils are more susceptible to NO_3^- leaching losses (Burkitt, 2014).

Moreover, New Zealand soils exhibit considerable natural variability, even among those classified as free draining, which can lead to notable differences in preferential flow and drainage patterns (Monaghan et al., 2007; Silva et al., 2000). Within a single paddock, soil profiles may differ due to changes in soil characteristics like permeability, resulting in inconsistent drainage. In addition, farm infrastructure and landscape features such as slope, gateways, fence lines, and water troughs may further influence the spatial variability of drainage across dairy pastures (Lucci et al., 2012).

2.6.2 Spatial and temporal variability

The spatial and temporal distribution of urine patches in pastures is complex due to the heterogeneity of urine distribution across the paddocks. These include the urine and non-urine patch areas, urine patches of different ages and different volumes and N concentrations. These variables are influenced by factors such as grazing dates, individual cow differences, feed variations, and daily fluctuations in urine volume due to water intake (Draganova et al., 2016; Ramírez, 2017).

The study of Moir et al. (2010) found that the mean annual area covered by urine patches was approximately 23% of the paddock area, with considerable variation between seasons and years. Additionally, Draganova et al. (2016) reported that urine patch distribution is influenced by cow behaviour and environmental factors, leading to accumulation in certain paddock areas rather than a homogeneous spread. They noted that 85% of urine was deposited on pasture, with 10% captured in holding yards and milking sheds. However, within paddock variations were not found important in determining urine distribution patterns in this study. Therefore, an adequate number of sampling points is necessary to accurately capture both the urine patches and the areas between them within a selected study site.

Moreover, drainage can alter over time due to changes in factors like rainfall, irrigation, and farm management practices, needing continuous monitoring to capture these dynamics (Carrick et al., 2013b). Due to this irregularity, determining the number of monitoring sites required to obtain a descriptive average of soil urinary N is challenging. High spatial variability suggests that a larger number of monitoring sites or measuring total N leaching over a larger area would provide a more precise measurement.

Given these challenges, both direct methods (e.g. lysimeters, water flux meters) and indirect methods (e.g. water balance, moisture sensors) have been used to quantify soil water drainage (Carrick et al., 2013b). According to previous literature, intact column lysimeters (Cameron et al., 1992), repacked lysimeters/flux meters (Norris et al., 2016), and ceramic suction cups (see section 2.6.3) have commonly been used in field leaching studies involving free-draining soils in NZ, whereas mole and tile drainage is widely used in poorly drained soils with a hard subsoil pan (Carrick et al., 2013b). Lysimeters can be re-packed or undisturbed soil columns within a plastic or concrete cylinder that allow drainage to be collected at the base. However, many types of smaller scale lysimeters can't be grazed, which restricts the ability to measure leaching under a grazed system (Burkitt, 2014). Although they are preferred for their ability to measure leachate from a known soil volume (Carrick et al., 2013b), the field installation of lysimeters is time-consuming and costly (Burkitt, 2014). Some lysimeters function as free-draining or under suction. Even though free draining lysimeters are less expensive and more convenient, there are major criticisms regarding restricted lateral flow. Moreover, they don't incorporate areas with high groundwater tables and can create saturated anaerobic conditions at the bottom of the lysimeter, which induces denitrification (Abdulkareem et al., 2015; Carrick et al., 2013b). Applying suction might reduce this challenge, yet this increases the cost when using at a large scale (Burkitt, 2014; Carrick et al., 2013b) and can lead to changes in the drainage characteristics and nutrient concentrations (Weihermuller et al., 2007). There is also an added

limitation for both small repacked lysimeters and ceramic suction cups, that draining water may bypass or be directed towards the device, influencing the accuracy of the measurements (Dennis, 2020b).

2.6.3 Suction cups

Ceramic suction cups are commonly used for soil water extraction in free-draining soils, particularly when larger volumes of samples are required in a relatively short period (Lord & Shepherd, 1993; Webster et al., 1993). Suction cups are commonly used to sample soil solution for NO_3^- and are installed at different depths (Grobler et al., 2003) and using different diameters (Surechem, 2024). Suction cups can be a cost-effective method of measuring drainage compared to lysimeters and can be installed in large arrays with low establishment costs (Dennis, 2020b). Also, the suction cups can be mounted easily with minimal disturbance to the soil profile, vertically, horizontally or on a 45° angle and are suitable for long-term monitoring due to their durability (Grosssmann & Udluft, 1991) (Figure 2.4). Vertical installed suction cups are useful for studying vertical movement and distribution of soil water at various depths. Horizontal installation is better for shallow soils and allows for minimal soil disturbance above the cup, which can help maintain natural soil water flow. It is also advantageous to install the suction cup at a slight angle, even in horizontal installation, to provide water penetration away from the pores cup. The installation of 45° angles further decreases the occurrence of preferential flow. Also, sealing the soil around the neck of suction cups using materials like sodium bentonite and silica flour can reduce preferential flow by enhancing hydraulic contact between the sampler and the surrounding soil (Curley et al., 2010).

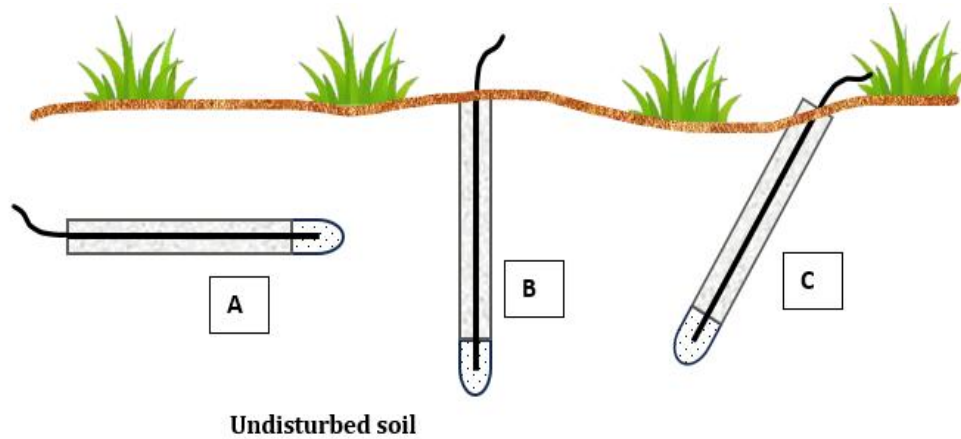


Figure 2.4: Horizontal installation of a suction cup (A), vertical installation (B), installation at a 45° angle (C), modified using Weihermuller et al. (2007)

The sample collection can be done at any time when the soil water potential is sufficient to allow extraction of the water from the soil (Grobler et al., 2003). Particularly, it's better to take samples soon after field capacity is achieved since the nutrient concentrations are often highest at this point (Lord & Shepherd, 1993). While sampling, a vacuum pressure is created inside the sampling system to extract the soil solution via the micropores of the suction cup. Generally, the amount of pressure created should be determined by considering the soil type, soil moisture content and the amount of water necessary for analysis (Weihermüller et al., 2005). When the capillary pressure in the suction cup is lower than the soil, the soil solution flows into the suction cup (Grossmann & Udluft, 1991).

In contrast to lysimeters, suction cups do not measure the leaching losses directly, as they only take spot samples of nutrient concentration from a precise depth (Dennis, 2020b; Wang et al., 2012). Therefore, the amount of NO_3^- -N leached should be calculated by multiplying the concentration by the drainage volume, estimated using a soil water balance for example (Wang et al., 2012; Webster et al., 1993).

There are several drawbacks to using suction cups, and one of the major issues is variability in measurement. Since the ceramic suction cups are smaller, they only sample a small area of soil and they can be bypassed by drainage water (Dennis, 2020b; Grosssmann & Udluft, 1991) increasing variability in measurements (Wang et al., 2012). Therefore, an abundance of suction cups must be used to achieve a more reliable result (Lilburne et al., 2012). The accuracy of the sampling data can also be influenced by soil type, disturbed soil contact, spatial variation of the paddock, and soil water changes associated with rainfall and preferential flow (Burkitt, 2014; Grosssmann & Udluft, 1991; Weihermuller et al., 2007). Samples need to be collected from the field manually after drainage events, which is a time-consuming process. These devices are difficult to install in gravelly soils and should be established at an adequate depth below the root zone to sample the drainage water accurately (Dennis, 2020b). Usually, they are installed in large numbers and samples are bulked to account for variation and reduce sample analysis costs. Wang et al. (2012) assessed the accuracy of lysimeters and ceramic suction cups in measuring NO_3^- -N leaching across three different NZ soils: Gorge silt loam, Mataura sandy loam, and Lismore stony silt loam. The results suggested that suction cups provided reliable data in the Mataura sandy loam, a freely draining alluvial soil. However, their effectiveness was lower in the structured Gorge soil and the Lismore topsoil, likely due to preferential flow pathways that the limited sampling area of suction cups failed to capture.

2.6.4 Piezometers

Piezometers are commonly used in groundwater studies to monitor water quality at specific depths. They are important in NO_3^- -N leaching monitoring to assess NO_3^- -N concentrations in groundwater. Piezometers provide detailed vertical profiles of NO_3^- -N concentrations. They can be used for repeated sampling over many years, supporting longitudinal studies of NO_3^- -N leaching without disturbing the surrounding environment. Piezometer data is often

complemented by soil moisture sensors, lysimeters, and modelling to get a fuller picture of NO_3^- -N leaching dynamics. Their design, installation, and careful interpretation of data are crucial for accurate monitoring (Chaulya & Prasad, 2016).

2.6.5 Trench lysimeters

Due to some of the challenges with traditional methods (lysimeters and suction cups) for measuring NO_3^- leaching in free-draining soil, an automated field lysimeter system, called a trench lysimeter, has recently been developed (Dennis, 2020b). A trench lysimeter is a leaching measurement system used in drainage studies that is simple to install and sample. Furthermore, this system can be installed below the root zone regardless of gravel and rocks (Dennis, 2020b), which is a limitation of repacked lysimeters and ceramic suction cups (Wang et al., 2012).

In contrast to conventional cylindrical lysimeters, trench lysimeters involve a trench constructed within the soil. It's often lined with waterproof materials to allow monitoring and collection of data on water and nutrient movement through a soil profile more naturally. Trench lysimeter systems (i.e. GroundTruth Lysimeter) consist of three principal components: a lysimeter, an autosampler (collects, measures and subsamples drainage water) and a server (organises this data and presents it to the user) (Dennis, 2020b).

These lysimeters are repacked lysimeters with the same functional system that Norris et al. (2016) described. The GroundTruth lysimeter is generally 10–20 m in length, 30–40 cm in width, and 70–100 cm in depth, with a surface area ranging from ~4 to 10 m². The base and sidewalls are sealed, ensuring that all water entering the lysimeter is fully collected. It also consists of a hanging passive wick made of fibreglass, that draws water from the entire lysimeter area, measures drainage volume (by filling and emptying using a container of known volume), electrical conductivity (EC), in real-time, and then sends it to a server for further

managing (Dennis, 2020b). A 1% flow-proportional subsample of the drainage is taken for laboratory analysis to measure nutrient concentrations.

Trench lysimeters are suitable for freely draining soils, particularly with a water table that stays at least one metre deep during the entire year (Dennis, 2020b). Moreover, leaching losses from traditional lysimeter/suction cup devices can be highly variable as they can measure remarkably high or low NO_3^- -N concentrations due to very local effects such as urine deposition. In contrast, the trench lysimeters incorporate both urine and non-urine areas, representing average NO_3^- -N concentrations associated with grazing across the measured area (Dennis, 2020b). Most importantly, if there is a manual sample data loss, the nutrient concentration of missing samples can be estimated using EC measurements. Therefore, researchers can assess nutrient loss even if they are unable to reach the site directly (Dennis, 2020b).

A simulation study compared the precision of GroundTruth lysimeters to commonly used flux meters and suction cups (Dennis, 2020b). Using 12 flux meters Norris et al. (2016) accurately measured nutrient leaching in 49% of years, a level achievable with a single GroundTruth lysimeter. Talbot (2016) achieved 80% accuracy with 64 suction cups, requiring only three GroundTruth lysimeters for equivalent precision. Moreover, greater accuracy can be achieved using a larger number of suction cups or lysimeters (96%) (Dennis, 2020b). For instance, a hypothetical intensive system of 144 suction cups, 3 m spacings, and 12 rows of 12 suction cups of lysimeters may meet the accuracy of an array of seven or eight GroundTruth lysimeters (Dennis, 2020b). Therefore, installing trench lysimeters significantly reduces ongoing in-field labour costs due to automatically measured drainage volumes and minimised analysis costs due to integrated event samples compared to other conventional methods (Dennis, 2020b).

2.7 Summary

As a result of this literature review and identification of research gaps, the following study tested the performance of trench lysimeters to measure drainage and NO_3^- -N leaching and compared these NO_3^- -N concentrations to traditional suction cups. Both methods were used to compare NO_3^- -N leaching loads under standard pastures under contemporary management and diverse pastures under both regenerative and contemporary management.

CHAPTER 3: RESEARCH DESIGN AND METHODS

3.1 Site description

The research was conducted at Massey University's Dairy One farm (40°22'31"S 175°36'58" E) bordering the Manawātū River on the edge of Palmerston North, New Zealand (Figure 3.1). The farm is 33 m above sea level, and the average annual long-term (2002-2023) rainfall is 1007 mm (NIWA, 2024). The average soil temperature (2002-2023, 10 cm depth) ranges from 8.6°C in July to 21.6°C in January (NIWA, 2024) (Weather station 21963, 40.381°S, 175.609°E).



Figure 3.1: Experimental location and trench lysimeters located within each treatment paddock of the Whenua Haumanu study at Dairy 1 farm, Palmerston North, New Zealand. Aerial photo was taken prior to the establishment of the pasture treatments.

3.2 Farmlet experimental design

The research site is part of the Whenua Haumanu research program established in April 2022. The broader study aims to explore the impact of standard and diverse pasture species and contemporary and regenerative management on a range of soil, pasture, animal, production and

environmental outcomes under a replicated dairy farmlet system. Three farmlets (12 ha each) were established with different pasture treatments namely, standard ryegrass-white clover pastures under contemporary management (Std-Con), diverse pastures under regenerative management (Div-Reg) and diverse pastures under contemporary management (Div-Con). The treatments were established in April 2022 with the direct drilling of different pasture species according to Table 3.1.

Table 3.1: Different pasture species sown in April 2022 according to treatment

Standard Pasture	Diverse Pasture	
Diploid perennial ryegrass	Diploid perennial ryegrass	Crimson clover
Tetraploid perennial ryegrass	Tetraploid perennial ryegrass	Vetch
Red Clover	Red Clover	Lucerne
White clover large leaved	White clover large leaved	Tall Fescue
White clover medium/large leaved	White clover medium/large leaved	Meadow fescue
	Chicory	Cocksfoot
	Plantain	Timothy
	Balansa clover	Phalaris
	Persian clover	Prairie grass

Each farmlet was stocked at slightly different stocking rates in the 2023 and 2024 drainage seasons to match pasture yields and ensure the grazing pressure required to achieve contemporary and regenerative grazing management (Table 3.2). The cow hours (Figure 4.6) were calculated by multiplying the number of cows grazing the treatment by the hours of grazing.

Table 3.2: Stocking rate for each treatment across the 2023 and 2024 drainage seasons.

Year	Stocking rate (cows/ha)		
	Std-Con	Div- Reg	Div- Con
2022/2023	2	2	2
2023/2024	2.5	2.5	2.5
2024/2025*	2.5	2	2

*This stocking rate only came into effect after calving in September 2024.

Contemporary management is described as having lower post-grazing residuals follows the DairyNZ best practice grazing guidelines and uses mineral/synthetic fertiliser and chemical sprays as required. Regenerative management is described as having longer grazing intervals, higher post-grazing residuals and lower mineral/synthetic fertiliser and chemical spray use. Details of N fertiliser application over the 2023/2024 drainage years are presented in Tables 3.3 and 3.4.

Each farmlet was managed as a system, which meant that cows were only grazed within their respective farmlets, only fed supplements harvested within the individual farmlet and milk production was measured from individual farmlets.

Table 3.3: Mineral N fertiliser applications in 2023

Treatment	Date and amount of N fertiliser applied (kg N/ha)			Total kg N/ha
	27/04/2023	01/5/2023	16/11/2023	
Std-Con	-	23 ^B	37 ^C	60
Div-Reg	11.5 ^A	-	9.2 ^C	20.7
Div-Con	-	23 ^B	37 ^C	60

^A Applied as urea (46% N), ^B Applied as N Protect (46% N), ^C Applied as granular Ammonium Sulphate (21% N)

Table 3.4: Mineral N fertiliser applications in 2024

Treatment	Date and amount of N fertiliser applied (kg N/ha)		Total kg N/ha
	05/03/2024	03/05/2024	
Std-Con	29.8 kg ^B	-	29.8
Div-Reg	-	15.2 ^A	15.2
Div-Con	29.8 ^B	30.4 ^A	60.2

^A Applied as urea (46% N), ^B Applied as N Protect (46% N)

In May 2024, a second application of N fertiliser was applied to the Div-Con treatment to stimulate pasture production (Table 3.4), as a result of an apparent deficiency of soil N which resulted in the pasture yield not meeting the dietary requirements of animals. Given this is a farmlet study and animals can only graze their allocated treatments, any deficiency in pasture production needs to be addressed through N fertiliser application or through supplementary feed (which also introduces N into the farmlet) to meet the project's animal ethics obligations. This difference in N application rate between the Std-Con and Div-Con treatments has been taken into account with the interpretation of results.

3.3 Soil properties

The research site consisted of free-draining Alluvial soils (Manawatū Sandy Loam), with soil properties summarised in Table 3.5 (S-MapOnline, 2024).

Table 3.5: Key soil properties of Manawatū Sandy loam soil based on Smap fact sheet (S-MapOnline, 2024)

Local soil name	Manawatū sandy loam
Smap sibling name	Awatere_24a.1 (Awate_24a.1)
Soil order/ classification	Weathered Fluvial Recent Soil
Profile texture	Topsoil: typically has a sand texture and is stoneless. Subsoil: dominantly sand textures with less than 3% gravel content. The plant rooting depth extends beyond 1m.
Drainage class	Well drained
Profile available water (PAW) (0- 60cm)	High (91 mm)
N leaching potential	Moderate
Vulnerability to waterlogging	Very low

Source: <https://smap.landcareresearch.co.nz/maps-and-tools/factsheets>

3.3.1 Soil physical properties

Key soil physical properties were measured from each treatment paddock (Table 3.6). Bulk density was measured in 2023 and 2024 by pushing cores (5 cm diameter and 5 cm depth) into the soil at 0-5 and 5-10 cm soil depths, carefully excavating the cores and then drying them at 105°C for 48h and then weighing to calculate the volume of soil in g/m³. Three sets of each core depth were sampled along a set transect for each treatment paddock.

The readily available water (RAW), total available water (TAW) and macroporosity were measured in 2023 by pushing three sets of each duplicate core (5cm diameter and 2 cm depth) into the soil at 7-9 cm and 13-15 cm depths. Depths were sampled along a set transect for each treatment paddock. One set of cores was placed on a ceramic plate and pressure vessel at -10

kPa (field capacity) and the second set were placed at -100 kPa (stress point) for ~ 1 week. A handful of loose soil was collected from each sampling depth and location and placed on a -1500 kPa (permanent wilting point) pressure plate for ~ 2 weeks. After which each core was weighed, dried at 105°C for 24h and then weighed to calculate the gravimetric water content. The volumetric water content was calculated by multiplying the bulk density by the gravimetric water content. The volumetric water content of soil between the field capacity (-10 kPa) and stress point (-100 kPa) determined the RAW. The volumetric water content of soil between field capacity (-10 kPa) and permanent wilting point (-1500 kPa) determined the TAW. Macroporosity was calculated as the total porosity minus the volumetric water content measured at -10 kPa using the method of Klute (1986) and an assumed particle density of 2.6 g/cm³.

Soil infiltration was measured (2023 and 2024) in six positions along the transect by selecting a flat area in the paddock, where a ring with a sharpened edge was placed into the soil. Water was added to half the height of the ring and left for ~20 minutes and refilled as necessary to establish a steady flow. Once flow stabilised, water was topped up to 60 mm, and readings were taken at timed intervals, ensuring the water was refilled to 60 mm within 3 seconds of each reading. Water levels and times were recorded until three steady readings were obtained, after which the sampling was completed and the infiltration rate per hour was calculated.

Table 3.6: Mean measured soil physical properties from each treatment paddock at Dairy One

Soil property	Depth (cm)	Std-Con	Div-Reg	Div-Con
Bulk density (g/cm ³)	0-5	1.27	1.15	1.21
	5-10	1.34	1.31	1.41
RAW (% v/v)	7-9	11.56	14.23	5.27
	13-15	7.85	10.57	3.83
TAW (% v/v)	7-9	22.98	25.09	20.22
	13-15	21.15	21.66	23.59
Macroporosity (%)	7-9	5.49	4.69	5.70
	13-15	7.68	9.55	5.30
Infiltration capacity (mm/hr)		115	118	200

3.3.2 Soil fertility

A representative soil sample (0-7.5cm) was taken for each treatment paddock on 23 May 2023 and 28 May 2024, before the onset of the drainage season of each year, which was analysed by Hill Laboratories for a range of soil fertility parameters (Table 3.7). Soil pH was determined using a 1:2 (v/v) soil: water slurry, Olsen P was analysed by adding soil to a 0.5M sodium bicarbonate solution at a pH of 8.5 and the mixture was shaken for 30 min, after which the filtered solution was analysed colourimetrically using the molybdenum blue method. Sulphate sulphur was determined using potassium phosphate extraction followed by ion Chromatography while Potassium (K), Calcium (Ca), Magnesium (Mg) and Sodium (Na) were determined using neutral ammonium acetate extraction followed by ICP-OES and cation exchange capacity (CEC) was calculated (sum of K, Ca, Mg, Na). Other measurements included Potentially Available N (PAN) and Anaerobically Mineralisable N (AMN)

(Anaerobic incubation followed by extraction using 2M KCl followed by Berthelot colorimetry), Total N (Dumas combustion) and Hot Water Extractable Carbon (HWEC) (measured on dried and sieved (<2mm) and extracted with water at 80°C for 16 hours according to Ghani et al (2003), followed by IR detection for Non Purgeable Organic Carbon) (Table 3.7)

Table 3.7: Soil chemical fertility for each treatment paddock measured on 23 May 2023 and 28 May 2024 (0-7.5cm depth)

Soil Property	2023			2024		
	Std-Con	Div-Reg	Div-Con	Std-Con	Div-Reg	Div-Con
pH	6.1	6	6	5.9	6.1	5.8
Olsen Phosphorus (mg/L)	39	48	26	30	40	35
Sulphate Sulphur (mg/kg)	4	8	10	5	11	18
Potassium (MAF units)	17	18	11	9	19	10
Calcium (MAF units)	8	8	9	8	8	9
Magnesium (MAF units)	36	38	31	34	38	31
Potentially Available Nitrogen (kg/ha)	249	264	195	229	255	222
Anaerobically Mineralisable N ($\mu\text{g/g}$)	161	167	136	150	159	146
Total Nitrogen (%)	N/A	N/A	N/A	0.23	0.23	0.24
Hot Water Extractable Carbon (mg/kg)	1,738	1,576	1,693	1,517	1,615	1,813
CEC (me/100g)	12	12	13	12	12	13

3.4 Trench lysimeters

Four replicated trench lysimeters were installed within each treatment paddock (1.2 ha each) to measure drainage volumes and NO_3^- leaching (Figure 3.1) using the methodology of Dennis (2020). The trench lysimeters were established prior to pasture sowing in February 2022. The trench lysimeters drain a surface area ranging from 3.1 – 3.6 m^2 (drainage surface areas of individual lysimeters are provided in Appendix 1) and are installed at an average depth of 70cm from the soil surface. To facilitate sample collection, the depth of the trench is graded from 60-70cm (collection end) to 70-90cm. The collection chamber is situated at ~130cm depth (Figure 3.2).

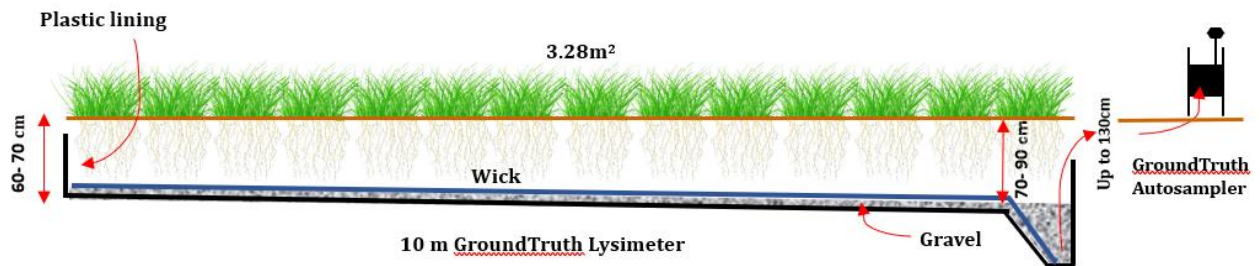


Figure 3.2: The GroundTruth lysimeter design used in the current study (Dennis, 2020b)

The method relied on a large passive wick strip (similar to a thick rope) made of fibreglass, which is 10 m long and 3-4 cm in diameter, surrounded by a layer of gravel. The entire trench was sealed with a plastic lining to ensure all drainage was captured (Dennis, 2020b). Once installed, the soil was carefully repacked in its original layers and was allowed to be equilibrated under grazing management for a period of 15 months prior to drainage collection (Figures 3.3). GroundTruth lysimeters were equipped with a flow meter to measure drainage volume and an autosampler to take a 1% flow-proportional subsample of drainage (Dennis, 2020b). The system used telemetry to alert the presence of drainage and the need to collect water samples. Samples were sent for laboratory analysis within 24 hrs of collection.



(a)



(b)



(c)



(d)



(e)



(f)

Figure 3.3: Images of installing the lysimeter (a) completed trench (b) plastic liner being inserted into a trench (c) wick covered with sand (d) finished lysimeter (e) final installed autosampler, inside (f) lysimeter after pasture establishment (Dennis, 2020a)

3.5 Suction cups

In order to validate the results collected from the GroundTruth lysimeter, a series of ceramic suction cups were installed between the trench lysimeters (Figure 3.4). Three rows of 10 suction cups were installed vertically in March 2024 at a depth of 50cm with individual cups within a row spaced approximately 1 m apart. Proximity of suction cup rows in relation to trench lysimeters is indicated in Figure 3.4. The vertical installation method was selected to avoid causing any damage to the existing vegetation as the pastures were already established.

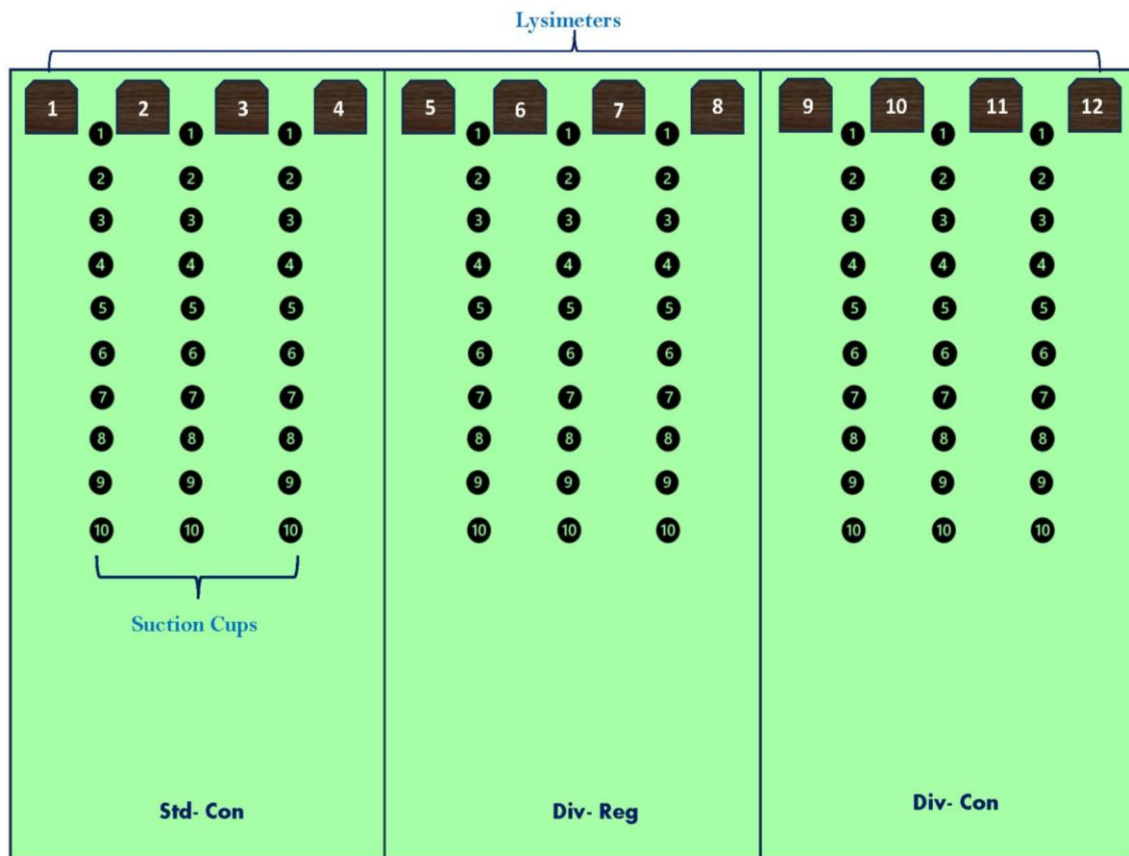


Figure 3.4: Suction cup placement in relation to the trench lysimeters (not to the scale).

Suction cups consist of a cylindrical ceramic cup (diameter ~2cm) with a microporous material fitted to a long pipe (Figure 3.5). They contain a smaller tube inside the pipe to collect the soil solution. The cups were installed using a hand-drilled hole of similar diameter. Before the ceramic cup was pushed into the hole, a slurry of soil extracted from the 45-50cm depth

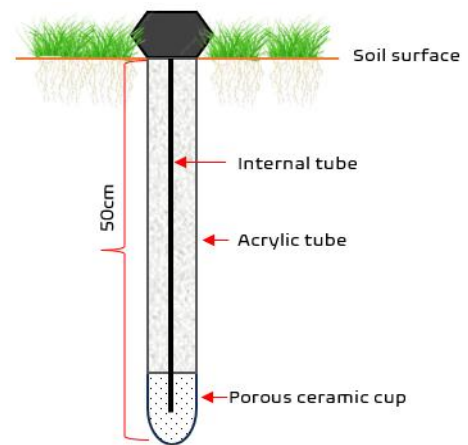


Figure 3.5: Ceramic suction cup

was placed into the hole to ensure constant capillary contact between the ceramic surface and the soil. A small amount of bentonite was applied on the soil surface around the neck of each suction cup to prevent preferential water flow (Grosssmann & Udluft, 1991; Webster et al., 1993). The suction cups each had a cut-off valve (Figure 3.6) to allow the cup to be placed under suction prior to sampling. The suction cups were allowed to equilibrate in the soil for 5 months before drainage was collected.

Prior to sampling, a partial vacuum of 50-70 kPa was created in each cup using a 60 ml plastic syringe connected to the tube via a three-way stopcock (Figure 3.6). After ~24 hrs, soil solution was withdrawn from the suction cups using the syringe and samples were transferred into separate containers, where individual volumes were measured. Soil solution volumes were highly varied for each suction cup. Therefore, nine composite samples were created by volume proportional mixing for each row (10 samples within one row into one composite sample), before being sent to the laboratory for analysis within 24 hrs of collection.

The sampling frequency was determined by the rainfall frequency to align with lysimeter drainage collections.



(a)



(b)



(c)



(d)

Figure 3.6: (a) Suction cup before installation (b) three-way stopcock (c) three-way stopper connected to the cut-off valve of the suction cup (d) plastic syringe (60ml) for creating vacuum and sample collection.

3.6 Nitrogen analysis

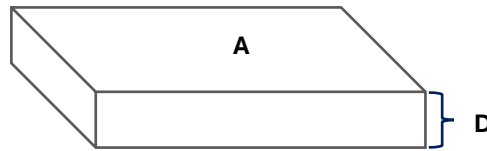
Drainage water samples from both the trench lysimeters and suction cups were filtered through a $0.45\ \mu\text{m}$ filter before analysis by the Central Environmental Laboratories, Palmerston North. Samples were analysed for nitrate-nitrogen (NO_3^- -N), nitrite-nitrogen (NO_2^- -N) and ammonium (NH_4^+ -N), and total soluble inorganic N (TIN) was calculated. NO_3^- -N and NO_2^- -N were measured using ion chromatography (APHA 24th Ed. 4110B), and NH_4^+ -N was measured using a spectrophotometer (modified APHA 24th Ed. 4500 NH_3 -F/ Phenate method).

3.7 Trench lysimeter nutrient loads

The leaching losses of each nutrient (NO_3^- -N, NO_2^- -N and NH_4^+ -N) were calculated using Equation 2.

D = Drainage depth

A = Area



Drainage volume (Water flux) (q_w) = Drainage depth (mm) \times Area (m^2)

Load (q_s) (kg/ha) = Water flux (q_w) (mm) \times Concentration (C) (g/m^3) \times 0.01 *Equation 2*

3.8 Climate data

Rainfall, soil temperature, evaporation and soil moisture data were sourced from the NIWA (National Institute of Water and Atmospheric Research) weather station 21963 (40.381°S, 175.609°E) which was situated about 850m away from the study site, at 39 m above sea level.

3.9 Soil water balance

A soil water balance was developed to compare against actual measured drainage and to estimate plant uptake and evaporation of soil water in 2023 and 2024. At least, three key factors must be estimated to calculate a soil water balance; amount of rainfall, evapotranspiration, and available water holding capacity (AWHC), which represents the maximum volume of water the soil can store and provide to vegetation when rainfall is insufficient to meet the evaporative demand (Horne & Scotter, 2016).

The following equation (Equation 3) was used to estimate drainage volume, and the soil water balance was calculated by using the model developed Scotter et al. (1979).

$$\text{Drainage Volume} = (\text{Rainfall} + \text{Irrigation}) - (\text{Evapotranspiration (ET)} + \text{Runoff} + \Delta \text{ Soil Moisture}) \quad \text{Equation 3}$$

3.10 Statistical Analysis

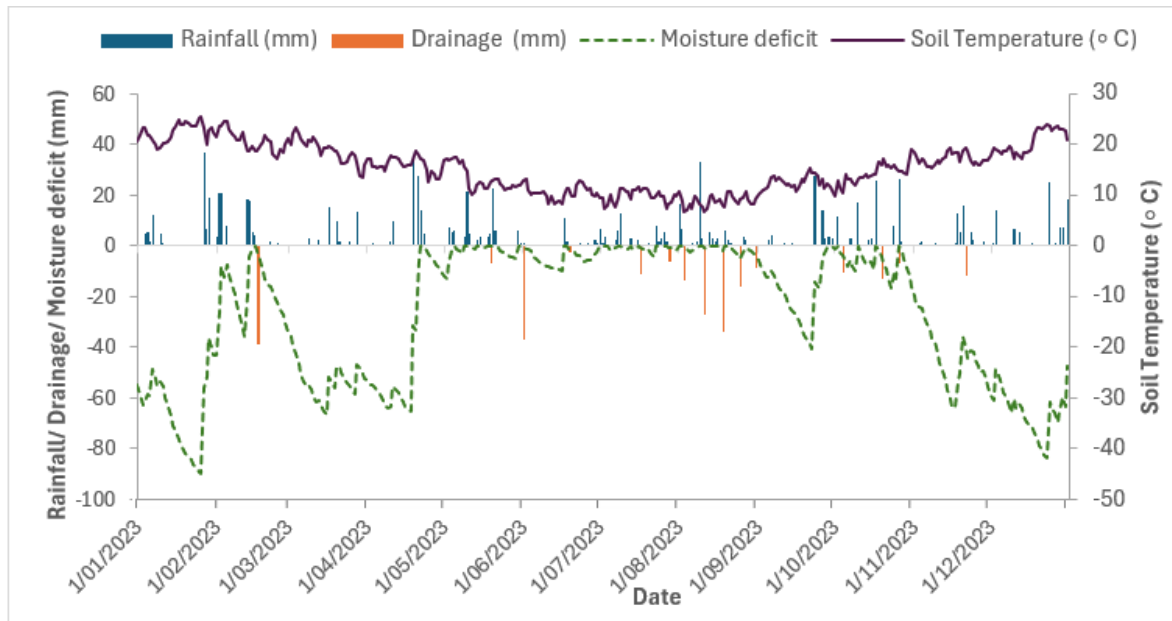
Data were analysed using the GLM procedure in SAS 9.4 GLM (SAS Institute Inc., Cary, NC, USA) and data visualisations were generated using Microsoft Excel. The NO_3^- -N concentrations and total NO_3^- -N load (kg N/ha) of drainage samples were analysed separately for each year using a model with the fixed effect of pasture treatments (Std-Con, Div-Reg and Div-Con) and drainage date. Drainage volume was fitted as a covariate. The effects of pasture treatment on cumulative drainage and cumulative NO_3^- -N load were compared using the least squares means test, and significance was declared at $P < 0.05$. The NO_2^- -N and NH_4^+ -N were not analysed due to very low concentrations. All models included four replicate samples per analysis period (drainage date) to represent the separate lysimeters.

CHAPTER 4: RESULTS AND DISCUSSION

4.1 Rainfall and drainage

The annual rainfall for the 2023 and 2024 study years was similar at 927 and 928 mm/yr, respectively, but this was lower than the 21-year average (2002-2023) of 1007 mm for the nearest weather station (NIWA weather station 21963). The highest long-term average monthly rainfall (2002-2023) at the site was in June (100 mm), with rainfall most frequent in winter and spring. In 2023, rainfall occurred frequently throughout the year, with several peak rainfall events. Total monthly rainfall was notably higher in January (94mm), May (101mm) and August (102mm). The highest total monthly drainage (100mm) was detected in August and continued in response to rainfall until mid-November (Figure 4.1). Conversely, in 2024, rainfall was more concentrated over the late winter and spring months, and drainage did not begin until mid-August, when the highest monthly rainfall was measured (210mm). A second peak in rainfall and most of the drainage occurred in September (142mm), with drainage continuing until 18th Nov 2024 (Figure 4.1).

(a) 2023



(b) 2024

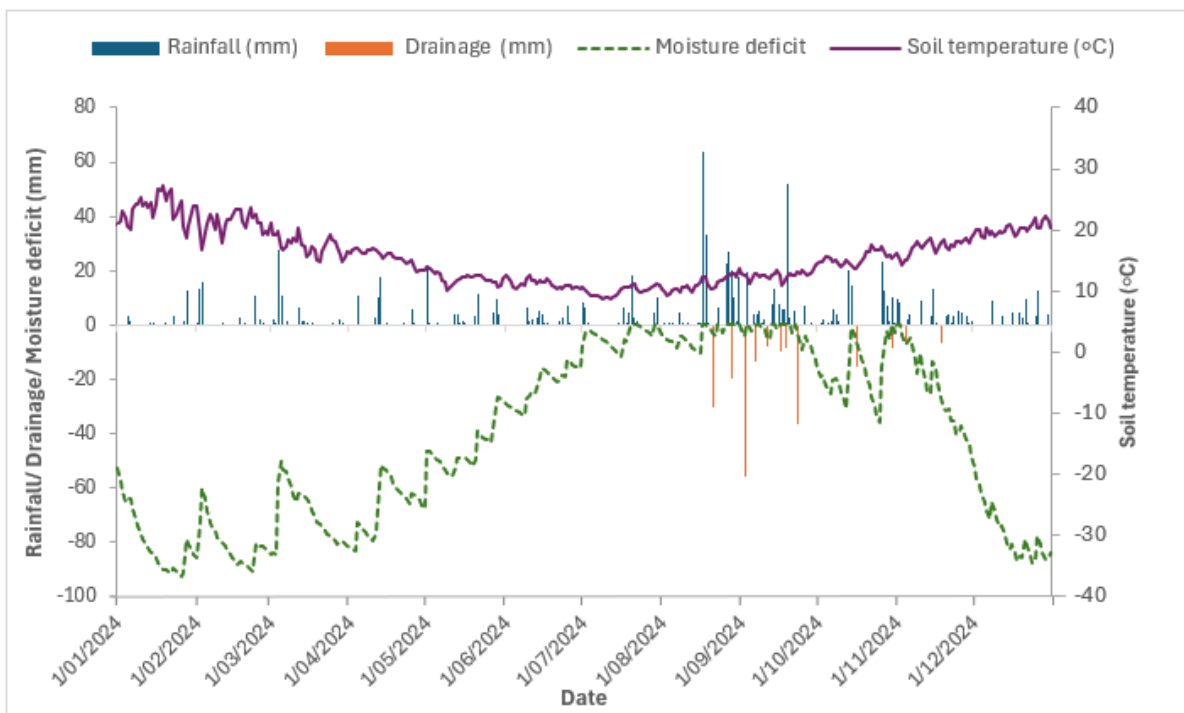
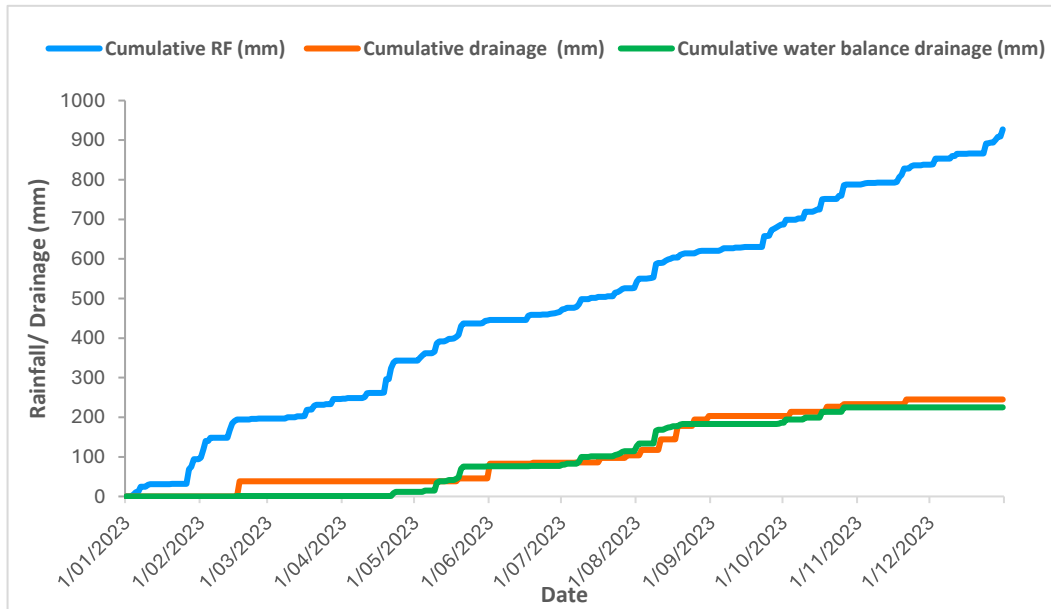


Figure 4.1: Mean drainage, daily rainfall, soil temperature and modelled soil water deficit for 2023–2024. Rainfall (mm) and soil temperature at 10cm depth were measured at the NIWA weather station 21963, and mean drainage depth (mm) was measured via trench lysimeters at Massey Dairy One farm, Palmerston North. Although soil water deficit and drainage are positive values, they were displayed as negative values for the purposes of this figure.

4.1.1 Cumulative rainfall and drainage

The cumulative drainage measured in 2023 and 2024 was 245mm and 220mm, respectively, compared to cumulative rainfall of 927mm and 928mm (Figure 4.2).

(a) 2023



(b) 2024

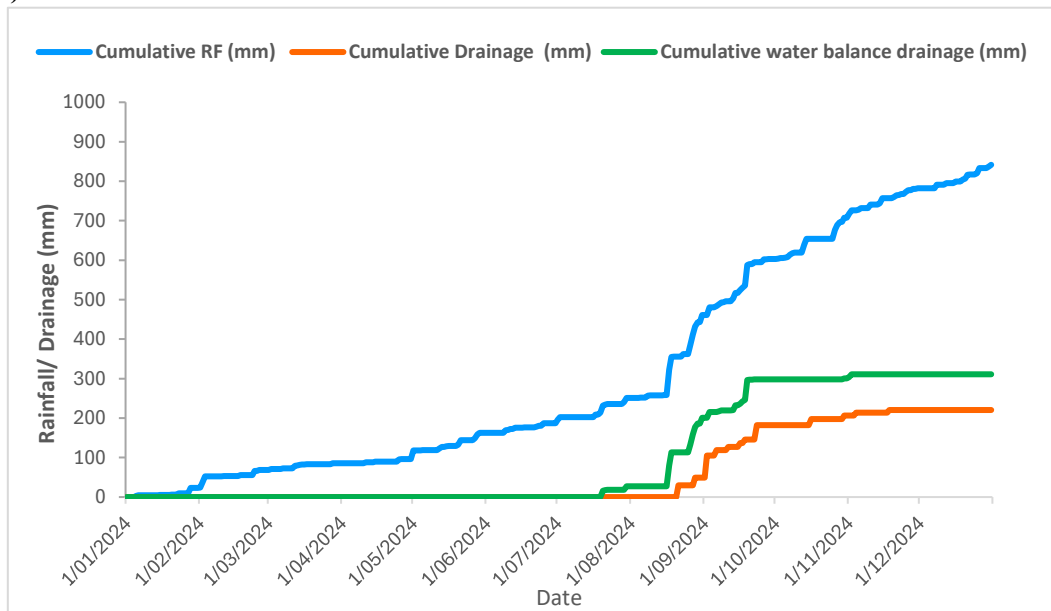


Figure 4.2: Mean daily rainfall (mm) and cumulative drainage depth (mm) measured via trench lysimeters at Massey University Dairy One farm for all treatments and modelled water balance drainage in (a) 2023 and (b) 2024.

Drainage generally corresponded with periods of zero soil water deficit, which was predicted using a soil water balance based on Scotter et al. (1979). Cumulative drainage increased sharply in response to the largest rainfall events in both years. However, there were also instances where rainfall did not result in significant drainage, possibly due to an existing soil water deficit.

The water balance model estimated total drainage volumes of 224 mm in 2023 and 310 mm in 2024. A discrepancy was observed between the drainage predicted by the water balance model and that measured using trench lysimeters in 2023 (Figure 4.2). This divergence was attributed to Cyclone Gabrielle (12–15 February 2023), which generated substantial rainfall in the upper catchment but not locally at the research site, leading to flooding of the nearby Turitea Stream. Consequently, surface water inundated part of the lysimeter site, resulting in 106 mm of additional drainage recorded exclusively in the Std-Con treatment. As this surface flooding was not generated by local precipitation, it could not be accounted for in the water balance model. Nonetheless, the measured drainage was included in the dataset as it contributed to actual drainage events and associated nutrient losses (Figure 4.3). In contrast to other leaching studies (Maheswaran et al., 2022), the difference between actual and predicted drainage is pronounced in 2024, and the cause of this discrepancy is unclear. The water balance provides an indirect estimation of drainage and the risk of NO_3^- -N leaching, with accuracy dependent on the precise measurement of various hydrological components. Therefore, more detailed soil hydraulic measurements may be required to improve these predictions.

4.1.2 Cumulative drainage comparison between treatments

2023

In 2023, the highest cumulative drainage was recorded in the Std-Con treatment (303 mm), followed by Div-Con treatment (236 mm), while the lowest drainage was observed in the Div-Reg treatment (172 mm). Although the Std-Con treatment measured a significantly higher mean drainage depth than the Div-Reg treatment ($p = 0.027$), this elevated drainage was likely an artifact of the surface flooding that occurred as a result of Cyclone Gabrielle (as detailed in Section 4.1.1). This flooding contributed an additional 106 mm of drainage, substantially increasing the total annual drainage recorded for Std-Con treatment. Following this event, drainage likely occurred more readily in response to subsequent rainfall in these lysimeters due to the soil being saturated (i.e., zero water deficit) from the initial surface flooding event (Figure 4.3).

2023

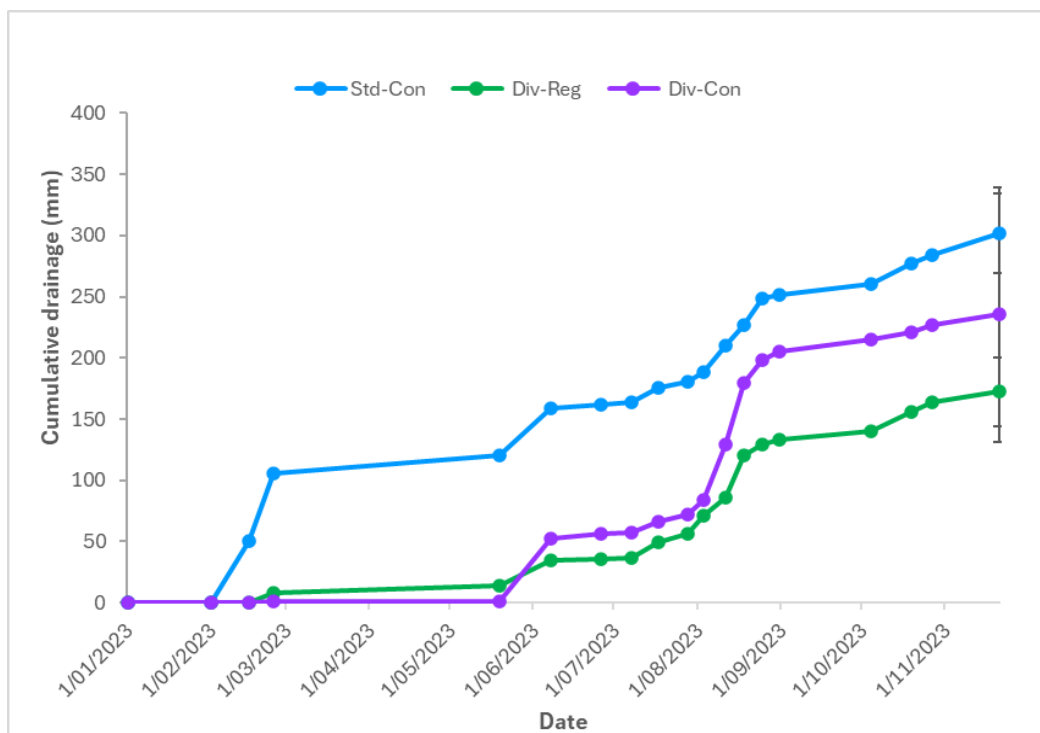


Figure 4.3: Cumulative drainage (mm) measured via trench lysimeters in each treatment 2023. Error bars represent the cumulative standard error of each treatment.

Only 8mm and 0.75 mm of drainage were measured in February for the Div-Reg and Div-Con treatments, suggesting that if the effect of the surface flooding event on drainage under the Std-Con treatment was removed, then total drainage depths between treatments after 1 May 2023 would be more comparable (Table 4.1). The majority of drainage across all the treatments occurred in the late autumn/winter period, making up 74%, 72% and 86% of the total drainage (excluding the drainage event associated with surface flooding) for the Std-Con, Div-Reg and Div-Con treatments, respectively (Table 4.1).

Table 4.1: Analysis of the seasonal and total measured drainage (mm) across the 2023 and 2024 drainage seasons

Year	Treatment	Late autumn/winter drainage (mm)	Spring drainage (mm)	Total drainage after 1 May (mm)	Total annual drainage (mm)
2023	Std-Con	146	51	197 ^A	303
	Div-Reg	125	47	172	172
	Div-Con	204	32	236	236
2024	Std-Con	44	163	207 ^B	207
	Div-Reg	58	153	211 ^B	211
	Div-Con	46	197	243 ^B	243

^A Excludes the drainage event associated with surface flooding. ^B This is the same as the total annual drainage of 2024 because there was no drainage before the 1st of May in 2024.

2024

In 2024, more uniform drainage was measured between treatments (Figure 4.4). Both the Std-Con and Div-Reg treatments measured similar cumulative drainage depths (~210mm) in 2024, whereas the Div-Con treatment recorded a slightly higher drainage depth of 243 mm, but there was no significant difference between any of the treatments ($p > 0.05$). The drainage did not

occur before 1 May 2024. In contrast to 2023, a higher proportion of drainage occurred in spring 2024, making up 78%, 72% and 81% of the total drainage for Std-Con, Div-Reg and Div-Con treatments, respectively (Table 4.1).

2024

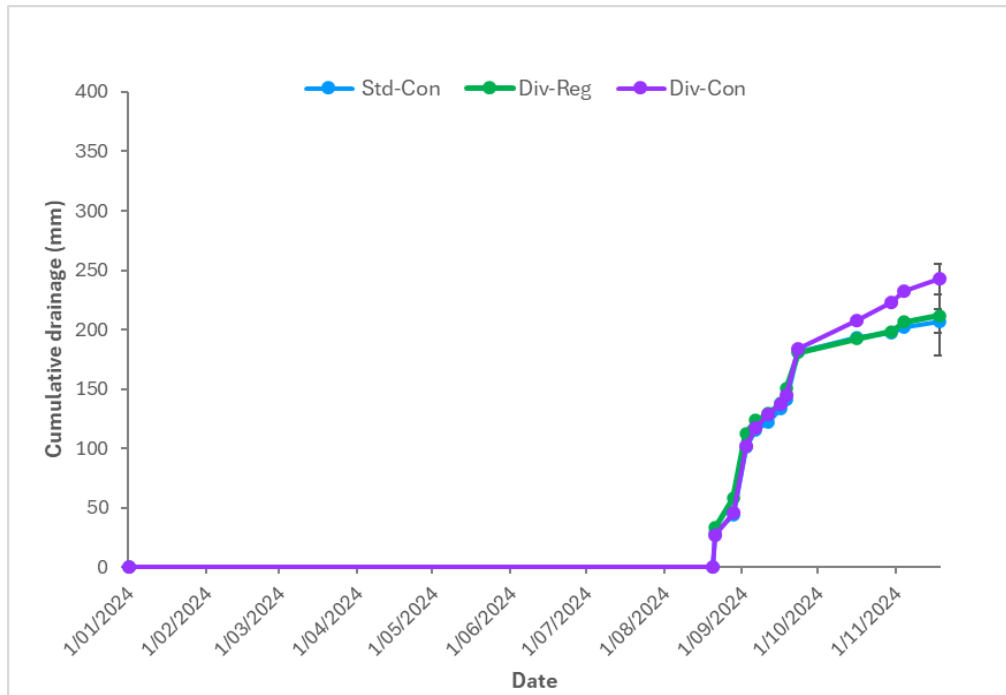


Figure 4.4: Cumulative drainage (mm) measured via trench lysimeters in each treatment 2024. Error bars represent the cumulative standard error of each treatment.

Given the reasonably consistent drainage depths across lysimeters and between treatments, this gives us confidence that the trench lysimeters are performing well in terms of measuring drainage. This is also supported by low standard errors around the drainage depths of all the treatments over the two years (Figures 4.3, 4.4).

4.2 Comparison of nutrient concentration between treatments

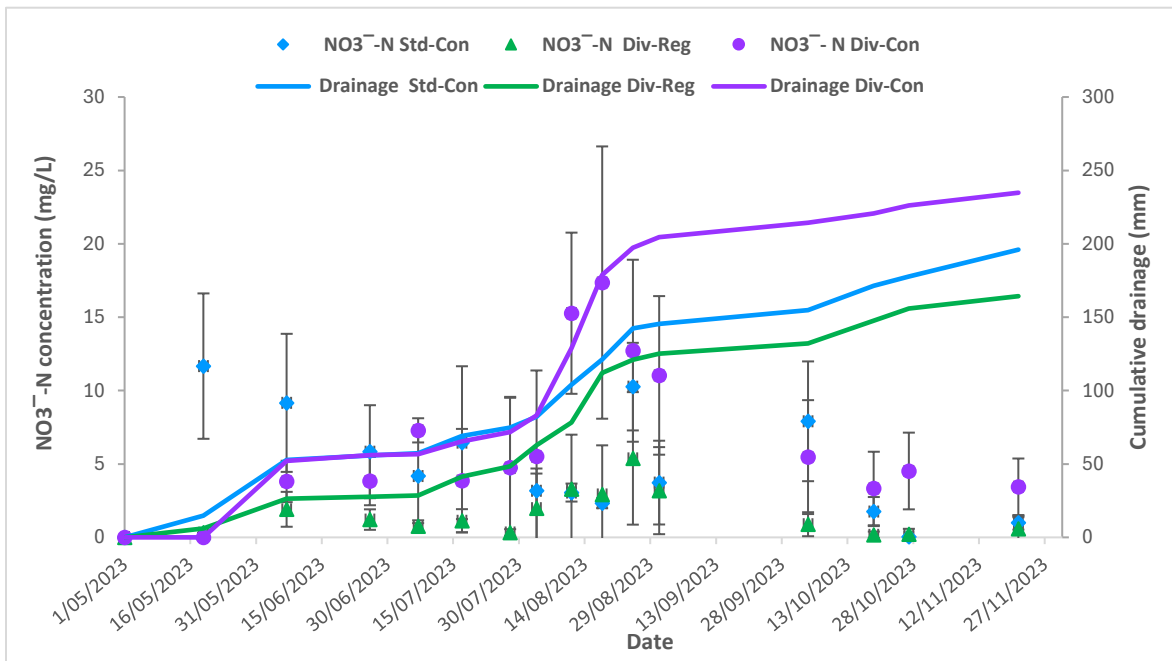
Although the NO_3^- -N, NO_2^- -N and NH_4^+ -N concentrations were measured in all water samples, NO_2^- -N and NH_4^+ -N concentrations were generally very low (< 0.06 mg/L) in both years. Therefore, only NO_3^- -N results are presented in this thesis.

4.2.1 Comparison of NO_3^- -N concentrations

Over both years, the Div-Con treatment recorded the highest NO_3^- -N concentrations, followed by the Std-Con treatment. In contrast, the Div-Reg treatment consistently maintained the lowest concentrations throughout the study period (Figure 4.5).

Elevated NO_3^- -N concentrations were measured in some lysimeters across both years. In 2023, lysimeter number 2 (Std-Con), and lysimeter numbers 10 and 12 (Div-Con) measured higher NO_3^- -N concentrations in August with mean August concentrations of 16.1, 10.9 and 30.0 mg/L, respectively, in comparison to the mean August concentration measured by the other lysimeters for these two particular treatments of only 2.1 mg/L (Std-Con) and 4.1 mg/L (Div-Con) (data not presented but can be made available on request). In 2024, lysimeters 1 (Std-Con), 10 and 12 (Div-Con) measured elevated NO_3^- -N concentrations in August and September events, with mean concentrations for these two months being 24.2, 31.2 and 36.4 mg/L for the three lysimeters, respectively (data not presented but can be made available on request). This contrasted with August and September mean concentrations measured by the other lysimeters under these treatments of only 8.0 mg/L (Std-Con) and 6.7 mg/L (Div-Con).

(a) 2023



(b) 2024

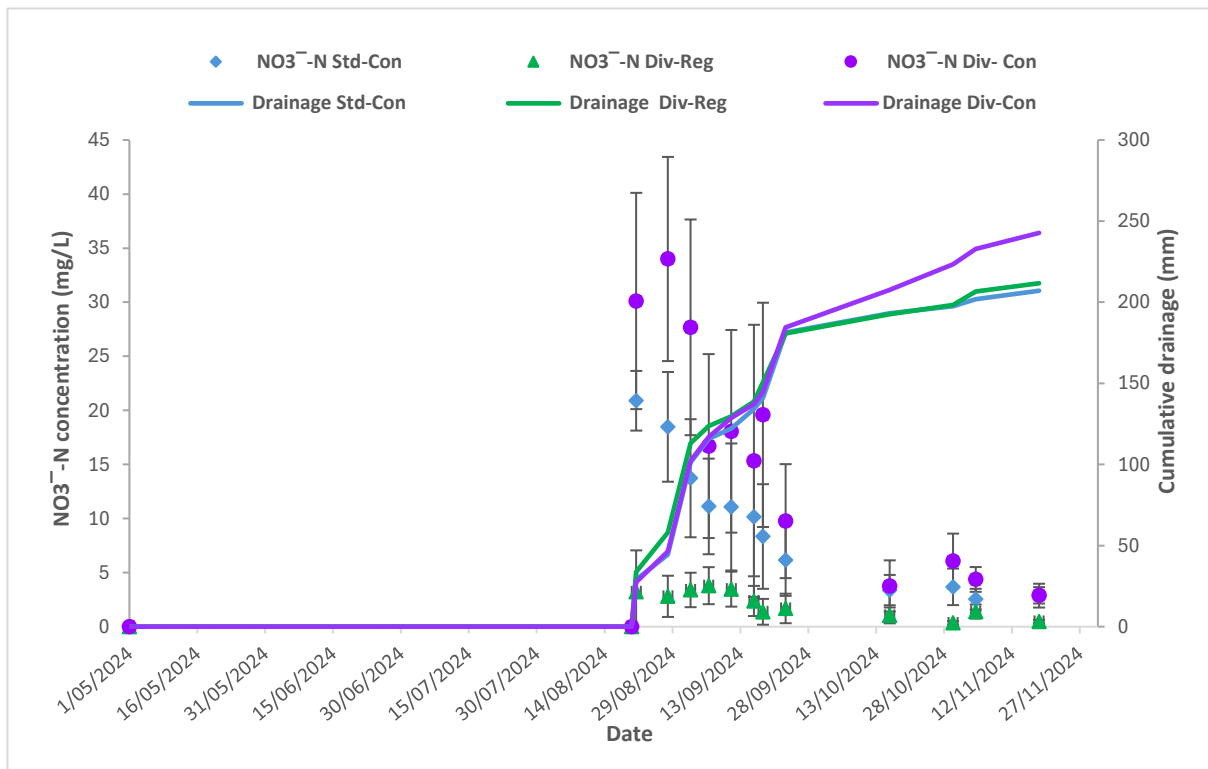


Figure 4.5: NO₃⁻-N concentration (mg/L) and cumulative drainage (mm) measured from trench lysimeters in (a) 2023 and (b) 2024, from Std-Con, Div- Reg and Div-Con treatments. Note that the 2023 drainage graph excludes drainage prior to 1 May, as this drainage was associated with Cyclone Gabrielle. Error bars represent the standard error values of each sampling event.

2023

In 2023, the mean annual NO_3^- -N concentrations in drainage water were 1.9, 5.8, and 7.8 mg/L for the Div-Reg, Std-Con and Div-Con treatments, respectively. During the first half of the winter drainage season (May to July), NO_3^- -N concentrations ranged from 0.3-1.9 mg/L, 4.1-11.6 mg /L and 3.8-7.3 mg /L for the three treatments, respectively (Figure 4.5). There was an increase in NO_3^- -N concentrations for all treatments in August, peaking at 5 and 10 mg/L for the Div-Reg and Std-Con treatments, respectively, with the highest concentration measured under the Div-Con treatment at 17 mg/L. Subsequently, a general decline in NO_3^- -N concentrations was observed, with final concentrations recorded at the last drainage event in November being 0.59, 0.99, and 3.45 mg/L for Div-Reg, Std-Con and Div-Con, respectively, a finding that is consistent with a number of other grazing studies (Christensen et al., 2019a; Monaghan et al., 2007; Talbot et al., 2021; Welten et al., 2014).

The higher NO_3^- -N concentrations measured under the Std-Con treatment in May could have been the result of the higher urine N concentration (measured as part of the wider Whenua Haumanu study, see Table 4.2) and high cow grazing hours in the late summer and early autumn of 2023 (Figure 4.6). The Div-Reg treatment measured the lowest NO_3^- -N concentrations, possibly due to lower cow grazing hours prior to the August drainage event (see Table 4.3 and Figure 4.6). This was combined with lower urinary N concentrations measured in summer and autumn (Table 4.2), which is the critical period of urinary N accumulation in soil that contributes to significant NO_3^- leaching in winter (Shepherd et al., 2011). The Div-Con treatment measured significantly higher NO_3^- -N concentrations ($p < 0.05$) than the other two treatments in 2023, peaking in August (Figure 4.5). However, this significance may be caused by the high NO_3^- -N concentrations measured in lysimeters 10 and 12 in August (discussed in section 4.2.1). A shelter belt was located ~25m from lysimeter 12. Therefore, it is possible that the cows spent more time in the vicinity of the shelter belt during the wet and cold conditions

in this August event and deposited more urine over lysimeter 12. However, it is difficult to explain why this feature may have influenced lysimeter 10 while having a lesser effect on lysimeter 11. Furthermore, a grazing event (Figure 4.6) that coincided with the highest rainfall event (Figure 4.1) could have been another reason for these elevated NO_3^- -N concentrations under Div-Con treatment in August.

Table 4.2: Dairy cow urinary N concentrations (%) measured in 2023 and 2024 as part of the wider Whenua Haumanu study

Date	% of N in Urine		
	Std-Con	Div-Reg	Div-Con
2023			
21/02/2023	0.491	0.273	0.354
18/05/2023	0.371	0.200	0.248
17/08/2023	0.247	0.296	0.273
12/10/2023	0.744	0.549	0.189
2/11/2023	0.253	0.163	0.270
12/12/2023	0.526	0.386	0.169
2024			
15/02/2024	0.498	0.314	0.365
16/04/2024	0.352	0.378	0.263
12/08/2024	0.346	0.313	0.533
16/10/2024	0.398	0.310	0.406
21/10/2024	0.279	0.271	0.265

2024

In 2024, the mean annual NO_3^- -N concentrations in drainage water were 2.2, 11.6 and 20.7 mg/L for the Div-Reg, Std-Con, and Div-Con treatments, respectively, with the Div-Con treatment being significantly higher than the other two treatments ($p < 0.05$). However, the elevated NO_3^- -N concentrations measured under the Div-Con treatment were likely due to the

issues identified earlier with lysimeters 10 and 12 (see the discussion above) and need to be interpreted with caution.

The drainage season commenced late, with the first drainage event occurring in mid-August 2024. The peak NO_3^- -N concentrations were 3.8, 21, and 33 mg/L for the Div-Reg, Std-Con and Div-Con treatments, respectively (Figure 4.5), and the highest NO_3^- -N concentrations occurred in late August for the Std-Con and Div-Con treatments and in early September for the Div-Reg treatment.

The NO_3^- -N concentrations gradually declined towards late spring across all treatments (Figure 4.5), with NO_3^- -N concentrations in drainage water at the last drainage event in November of 0.5, 2.8 and 2.9 mg/L for the Div-Reg, Std-Con, and Div-Con treatments, respectively. This general trend was also observed in 2023 and in previous studies (Christensen et al., 2019a; Monaghan et al., 2007; Talbot et al., 2021; Welten et al., 2014).

Differences of NO_3^- -N concentrations between the Div-Con and Std-Con may have also been influenced by the winter grazing. For instance, the Std-Con paddock was grazed earlier, in June, which would have allowed more time for N to be removed from urine patches via plant uptake and denitrification before the start of the drainage season in August, compared to the Div-Con treatment. Furthermore, in 2024, the Div-Con treatment received an additional N fertiliser application of 30 kg/N ha in May (Table 3.4) to stimulate pasture production due to pasture yield not meeting the cows' dietary requirements. This additional application of fertiliser might have contributed to higher cow N intake and return, resulting in the elevated urinary N measured in August, which may have led to the higher concentrations of NO_3^- -N measured in drainage under the Div-Con treatment.

The NO_3^- -N concentrations measured under the Div-Reg treatment were significantly lower compared to Std-Con treatment in 2024 ($p=0.003$). This result is likely influenced by the lower

N fertiliser inputs under regenerative management. Moreover, the Div-Reg pasture included species such as chicory and plantain, which are known to reduce the risk of NO_3^- -N leaching due to their higher water content (Marshall et al., 2021; Nguyen et al., 2022) and natural biological nitrification inhibitors (Gardiner et al., 2020), which is in line with the likely impacts discussed by (Grelet et al., 2021). Interestingly, the grazing records suggest that in 2024, the Div-Reg treatment had more cow hours than the Std-Con treatment in late summer and autumn before drainage commenced and remained higher until the end of the drainage period (Table 4.3; Figure 4.6). A comparison of the legume content between the two treatments revealed that the Std-Con treatment had a slightly higher legume content than the Div-Reg treatment before drainage, which may have also contributed to elevated NO_3^- -N concentrations in the Std-Con treatment (Appendix 2). Due to higher N concentration than grasses, higher legume content is likely to result in a higher N intake and return in urine patches (Peoples et al., 2009).

Table 4.3: Grazing cow hours recorded for each treatment for 2023 and 2024.

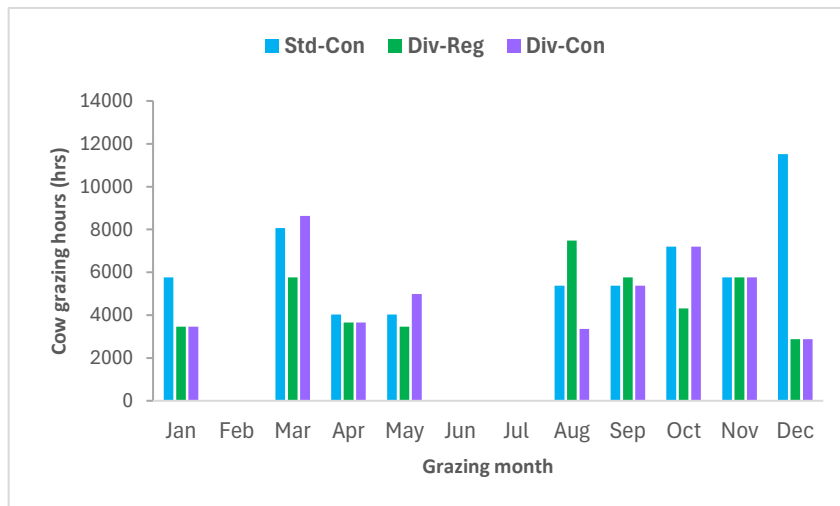
Treatment	Total cow grazing hours from 1 Jan to 1 Aug^A	Total grazing cow hours/year
2023		
Std-Con	21,888	57,120
Div-Reg	16,320	42,528
Div-Con	20,736	45,312
2024		
Std-Con	16,968	28,488
Div-Reg	27,072	33,984
Div-Con	23,760	32,976

^A For 2023, there were no grazings in June or July.

These high NO_3^- -N concentrations measured in early drainage events were likely sourced from urine patches deposited over late summer and autumn due to relatively low rainfall and reduced

plant growth (Di & Cameron, 2002a; Selbie et al., 2015; Shepherd et al., 2011; Silva et al., 1999; Talbot et al., 2021), whereas lower NO_3^- -N concentrations measured in late spring are likely due to several factors including increased pasture growth and N uptake, depleted NO_3^- -N in the soil associated with urine due to previous drainage and decreasing drainage depths (Christensen et al., 2019a). Moreover, denitrification and immobilisation in warm, wet soils could also contribute to the low NO_3^- -N concentrations measured in drainage at that time of year (Christensen et al., 2019a; Luo et al., 2008; Saggar et al., 2004).

(a) 2023



(b) 2024

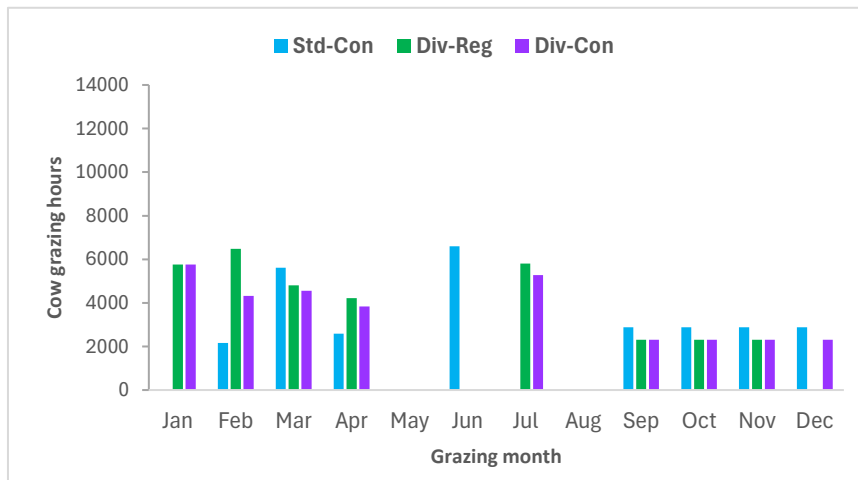


Figure 4.6: Grazing cow hours recorded for each month for Std-Con, Div-Reg and Div-Con treatments in (a) 2023 and (b) 2024.

The current study reported NO_3^- -N concentrations generally decreased when cumulative drainage depths reached about 150-200 mm (Figure 4.5) over both years. The variability in NO_3^- -N concentrations also decreased later in the drainage season and as drainage approached its maximum depth. This closely follows the typical seasonal trend of NO_3^- -N concentration in previous leaching studies (Christensen et al., 2019a; Houllbrooke et al., 2003; Malcolm et al., 2014; Monaghan et al., 2007; Wang et al., 2012).

4.3 Effect of treatment on NO_3^- -N loads

2023

The Div-Reg treatment exhibited the lowest cumulative annual NO_3^- -N load at 3 kg N/ha/yr, followed by the Std-Con treatment with a load of 14 kg N/ha/yr. In contrast, the Div-Con treatment resulted in a significantly higher NO_3^- -N load of 23 kg N/ha/yr compared to the other two treatments ($p < 0.05$) (Table 4.4). However, caution needs to be taken in reference to this significant effect due to the elevated NO_3^- -N concentrations measured in lysimeters 10 and 12 under the Div-Con treatment and the increased drainage measured under the Std-Con treatment due to surface flooding.

Table 4.4: Annual NO_3^- -N load measured in 2023 and 2024 for each treatment. Different letters show a significant difference between treatments ($p < 0.05$).

Treatment	NO_3^- -N load (kg/ha/yr)	
	2023	2024
Std -Con	13.67 ^b	24.11 ^b
Div - Reg	3.3 ^b	5.42 ^c
Div - Con	23.54 ^a	43.93 ^a

In 2023, the Std-Con and Div-Reg treatments showed a steady increase in cumulative NO_3^- -N load. The Div-Con treatment measured a sharp rise in NO_3^- -N leaching after early August (Figure 4.7). This was driven by the highest rainfall and resulting drainage (more than 100mm of drainage was measured from the Div-Con treatment within the first three August events), which also coincided with the higher NO_3^- -N concentrations due to the grazing event at the beginning of August (discussed in section 4.2.1), rather than an overall treatment effect over multiple grazing events.

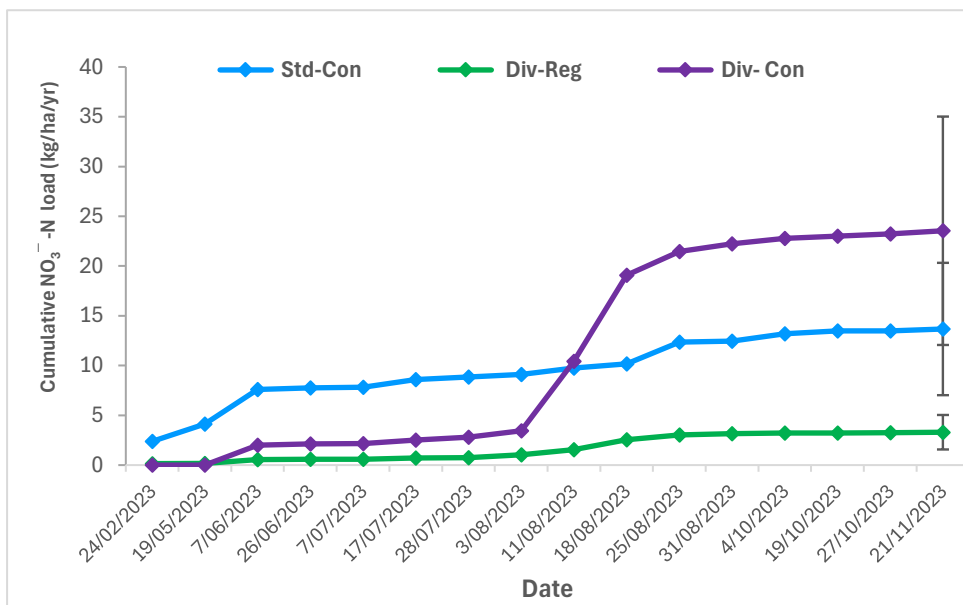


Figure 4.7: Cumulative NO_3^- -N loads (kg/ha) measured from each treatment (Std-Con, Div-Reg and Div-Con) in 2023. Error bars represent the cumulative standard error of each treatment.

The NO_3^- -N loads measured in 2023 in the current study under the Std-Con treatment on a well-drained sedimentary soil (14kg N/ha) were similar to the average NO_3^- -N loads published by Christensen et al. (2019a) under dairy grazing on a contrasting poorly drained Pallic soil (14kg N/ha). Houlbrooke et al. (2003) also recorded NO_3^- -N losses of 26 kg N/ha/year under dairy grazing on a Pallic soil (Tokomaru silt loam). In another study, Betteridge et al. (2007) reported NO_3^- -N leaching of 13 kg N/ha/year on a free-draining Oruanui sand under year-round

dairy grazing. Additionally, research on two high-rainfall dairy farms with different soil types in the Lake Rotorua catchment documented NO_3^- -N leaching losses on a free-draining pumice soil ranging from 20 to 42 kg N/ha, while a slower-draining podzolized pumice soil had lower annual N leaching losses, ranging from 15 to 22 kg N/ha (Sprosen & Ledgard, 2020).

2024

The NO_3^- -N loads were higher in 2024 at 5, 24 and 43 kg/ha for Std-Con, Div-Reg and Div-Con treatments, respectively (Table 4.4; Figure 4.8). The NO_3^- -N loads were significantly higher under the Div-Con treatment compared to the other two treatments ($p < 0.05$), and the Div-Reg treatment measured significantly lower loads than the Std-Con and Div-Con treatments ($p < 0.05$). The elevated NO_3^- -N leaching in 2024 compared to 2023 was mainly driven by the delayed onset of rainfall and drainage.

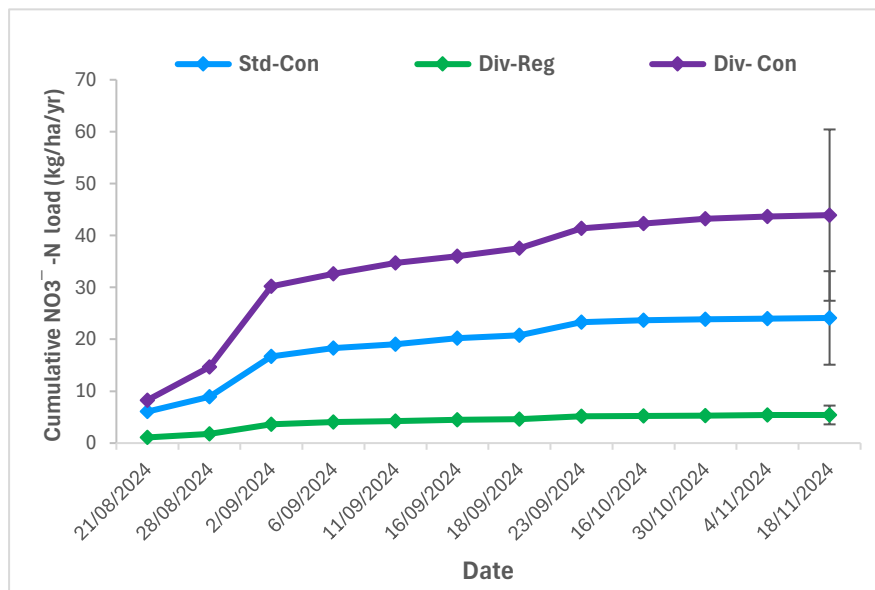


Figure 4.8: Cumulative NO_3^- -N loads (kg/ha) measured from each treatment (Std-Con, Div-Reg and Div-Con) in 2024. Error bars represent the cumulative standard error of each treatment.

In the current study, NO_3^- -N leaching loads under the Div-Reg treatment were only 21.4% and 20.8% of those measured under the Std-Con treatment in 2023 and 2024, respectively (Table 4.4), with low variability and standard errors around the measurements (Figures 4.7 and 4.8).

Possible reasons for reduced NO_3^- -N leaching under the Div-Reg treatment could be due to regenerative farm management practices that included lower N fertiliser inputs (Table 3.4) and increased pasture diversity resulting in lower NO_3^- -N leaching (Table 3.1), as pastures included species such as chicory (Cheng et al., 2015; Mangwe & Bryant, 2022) and plantain (Box et al., 2017; Marshall et al., 2021; Navarrete et al., 2018; Nguyen et al., 2022). These species are known to alter the concentration and frequency of urine patches and produce biological nitrification inhibitors (Gardiner et al., 2020). Other deep-rooted species, such as lucerne could be absorbing N from deeper soil layers (Beukes et al., 2014; Crush et al., 2005). Furthermore, drought-tolerant species such as tall fescue, phalaris, and cocksfoot, which grow under NZ's drier summer and early autumn conditions (Fulkerson & Donaghy, 2001), will be actively taking up N over this period. However, previous research has reported mixed findings regarding NO_3^- -N leaching diverse pastures. Some studies indicated the potential of diverse pastures to reduce N losses through the urine of lactating cows (Beukes et al., 2014; Edwards et al., 2015; Totty et al., 2013; Woodward et al., 2012), while others found no significant difference between standard and diverse pastures (Graham et al., 2024).

Over both years, the current study questioned the higher NO_3^- -N concentrations and loads measured under the Div-Con treatment due to elevated NO_3^- -N concentrations measured in two of the four lysimeters (i.e., some lysimeters under the same treatment individually showed ~40-fold differences in concentrations in the same rainfall event). The high variability and, therefore, high standard errors (Figures 4.7 and 4.8) surrounding these measurements have reduced our confidence in the Div-Con results measured in the current study. Although these values are likely to be outliers, the persistently high concentrations across multiple events and years suggest that background soil N or an anomaly associated with the current grazing and urine deposition across these lysimeters resulted in higher mean NO_3^- -N concentrations from this treatment. One potential factor is the location of a shelter belt ~25m from lysimeter 12.

However, why this shelter belt may have influenced lysimeter 10 while having a lesser effect on lysimeter 11 remains unclear. Additionally, the site experiences strong winds at certain times of the year, which may have influenced cattle behaviour, leading cows to gather over those particular lysimeters, which measured higher NO_3^- -N concentrations, particularly in August and September (data not presented but can be made available on request). Therefore, there is a necessity for longer-term monitoring or the addition of more replication to clarify the leaching loads measured from the Div-Con treatment.

Nitrogen leaching from grazed pastures is a complicated process influenced by numerous factors, including climate conditions, soil characteristics, interactions between above- and below-ground organisms, and pastoral management plans (Vibart et al., 2016). Differences in plant distribution, soil characteristics, rainfall patterns, and temperature fluctuations all contribute to the variability in the amount of NO_3^- -N leached from each urine patch. Studies on NO_3^- -N leaching in free-draining soils are further challenged due to their inherent characteristics, the complexity of water movement, and the limited area that can be instrumented for drainage measurements (Lilburne et al., 2012). Plant roots, earthworm activity and soil wetting and drying can create surface-connected macropores within the soil profile which can either increase or decrease NO_3^- -N leaching rates (Cameron et al., 2013; Silva et al., 2000). Moreover, spatial and temporal variation in urinary N deposition can cause over or underestimation of NO_3^- -N leaching losses (Li et al., 2012; Selbie et al., 2015).

4.4 Suction cups vs lysimeters

Suction cup samples were taken only in 2024, and they generally measured much lower NO_3^- -N concentrations compared to trench lysimeters.

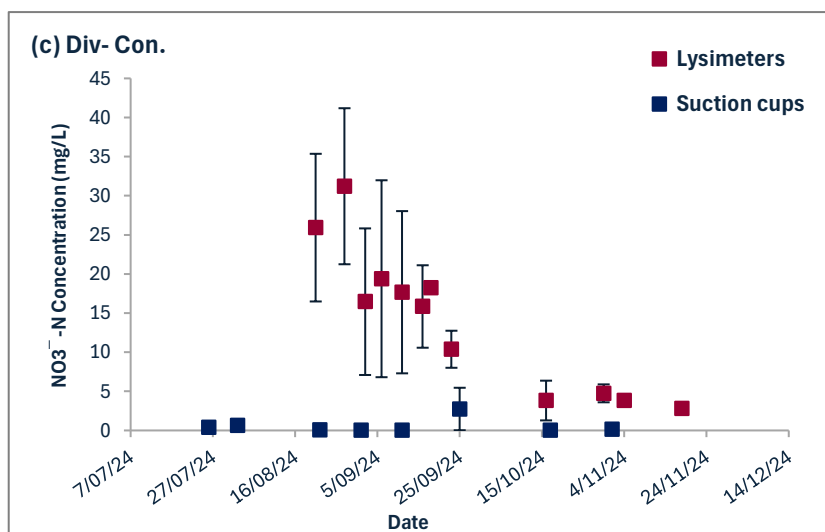
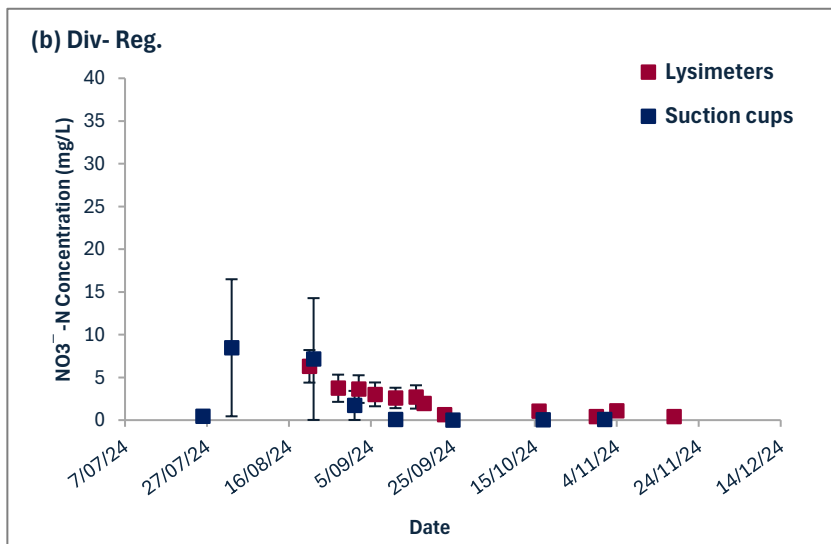
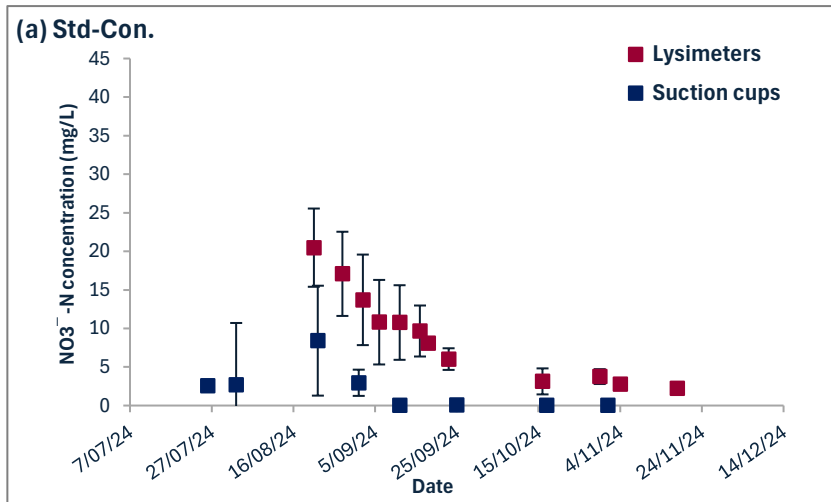


Figure 4.9: Mean NO_3^- -N concentrations measured via lysimeters and suction cups for each treatment in 2024 (a) Std-Con (b) Div-Reg (c) Div-Con. Error bars represent the standard error around the mean for each event.

Both suction cups and trench lysimeters showed a declining trend in NO_3^- -N concentrations (except suction cups under the Div-Con treatment) across the treatments as the drainage season progressed. In the context of these results, it is important to acknowledge the variability encountered with sampling (described in the methods) and the widely acknowledged limitations of suction cups (i.e. missing urine patches or areas of preferential flow) (Figure 4.9). Both methods facilitate the collection of soil water or leachate samples for NO_3^- -N concentration determination. However, trench lysimeters additionally provide an estimate of drainage quantity. In contrast, when suction cups are used, drainage quantity must be estimated separately using a soil water balance approach. (Carrick et al., 2013b; Dennis, 2020b; Weihermuller et al., 2007).

The trench lysimeters showed a reasonable level of accuracy in measuring drainage quantity as evidenced by low standard errors within each treatment (Figures 4.3 and 4.4). Compared to traditional lysimeters, which are difficult to install and generally measure a smaller surface area, this new automated trench lysimeter system is a more convenient method to measure drainage and collect representative samples on an event basis, as detailed in Chapter 2 (Literature review) of this thesis.

In contrast to trench lysimeters, individual suction cups can lead to miscalculation of NO_3^- -N concentrations depending on whether a suction cup is directly under a urine patch or not. In the current study, the suction cup results are more consistent when the concentrations are low, which could be when the suction cups are located where there has been no urine deposited. However, they are not very consistent in capturing NO_3^- -N leaching when concentrations are higher (i.e. some suction cups appear to have intercepted urine patches). In contrast, the trench lysimeters sample drainage from a larger surface area (Figure 3.2) and are designed to collect one integrated sample from each event. Due to this, they are likely to measure both urine patch and non-urine patch areas, urine patches of varying ages and concentrations, whereas suction

cups measure localised sampling areas. The difference between the methods is more pronounced in September, suggesting the trench lysimeters captured more variability in NO_3^- -N leaching dynamics, whereas suction cups appear to be less sensitive to these peaks, providing a more moderate estimate of NO_3^- -N concentrations. Therefore, a larger number of suction cups would be needed to account for the variability encountered with grazed pasture soils, as mentioned in previous studies (Lilburne et al., 2012; Wang et al., 2012).

We experimented with a free-draining alluvial soil, which showed similar results to the study of Wang et al. (2012), suggesting that while ceramic suction cups can be effective in measuring NO_3^- -N leaching in sandy loam soils with uniform texture, they may not provide accurate measurements in structured or stony soils due to preferential flow, whereas lysimeters are likely to be more reliable under these soil types. Modelling by Dennis (2020) suggested that 144 suction cups, arranged in a 12×12 grid with 3 m spacing, would be needed to achieve a level of accuracy comparable to an array of seven to eight GroundTruth lysimeters. Given that one sample would be generated from each trench lysimeter for each event, in contrast to 144 samples (or a subset of samples if groups of samples are bulked) using suction cups, this underscores the advantages of the trench lysimeter method.

Although the trench lysimeters are better than suction cups for estimating NO_3^- -N loads, the results of this study showed high variation between individual lysimeters within the same treatment (Table 4.5 and 4.6). Under the Div-Reg treatment, the measured loads were consistent with low standard errors between individual lysimeters and events (Figures 4.7 and 4.8, Appendices 3 and 4). However, other treatments showed larger differences in annual NO_3^- -N loads between lysimeters in some cases (i.e. ~20-50-fold difference under the Div-Con treatment). This suggests that factors influencing spatial urine deposition need to either be removed (i.e. prevent cows from using shelter belt) or urine deposition needs to be thoroughly characterised using visual observations or drone images which detect urine spots, post grazing,

to improve the interpretation of results. Additionally, increasing the number of lysimeters per treatment may be required to improve the accuracy of NO₃⁻-N leaching measurements in free-draining alluvial soils.

Table 4.5: Cumulative annual NO₃⁻-N load for each lysimeter 2023.

Treatment	Lysimeter	Cumulative NO₃⁻-N load (kg/ha)
Std-Con	1	18.01
	2	30.56
	3	3.79
	4	2.34
Div-Reg	5	0.18
	6	3.36
	7	1.56
	8	8.12
Div-Con	9	1.16
	10	52.40
	11	9.70
	12	30.92

Table 4.6: Cumulative annual NO₃⁻-N load for each lysimeter in 2024.

Treatment	Lysimeter	Cumulative NO₃⁻-N load (kg/ha)
Std-Con	1	50.10
	2	10.39
	3	22.18
	4	13.79
Div-Reg	5	4.27
	6	2.77
	7	3.88
	8	10.76
Div-Con	9	3.74
	10	75.81
	11	30.72
	12	65.44

CHAPTER 5: LIMITATIONS

On-farm monitoring of nutrient leaching from free-draining alluvial soils presents significant challenges. Currently available sampling devices, such as suction cups and lysimeters, have limitations in their suitability for measuring on-farm nutrient leaching in these soils. These limitations arise due to the higher spatial variability in soil types, drainage characteristics, management practices, and nutrient applications (i.e., urine patches) within a paddock. The establishment of future farm-scale leaching reference sites requires careful planning to ensure to address some of the experimental challenges experienced in the current study, such as spatial relevance, the effect of unusual grazing behaviour, the number of lysimeters/suction cups and adequate sampling replicates.

The trench lysimeter system utilised in this study presents certain limitations in measuring NO_3^- -N leaching, primarily due to the number of replicates and the restricted spatial coverage provided by the four lysimeters installed. However, increasing the number of lysimeters to enhance measurement accuracy is constrained by the cost of installation and the practical challenges of installation within a functioning dairy farm. For instance, the rotational lengths and cow grazing hours on each farmlet were dependent on cow dietary requirements and pasture growth.

The leaching of urine N across treatment paddocks varied due to multiple factors, including the volume of individual cow urine patches, the concentration of NO_3^- -N per patch, the age of the urine patches, prevailing weather conditions, seasonal influences or other unexpected constraints such as flooding. Additionally, the difference in grazing days (grazing did not occur simultaneously across all paddocks) between treatments likely contributed to differences in NO_3^- -N leaching rates and concentrations. This effect is further influenced by environmental

factors such as soil moisture and temperature, which can facilitate N losses through other processes like denitrification and volatilisation before leaching.

CHAPTER 6: CONCLUSION

The study shows the effectiveness of trench lysimeters over suction cups in measuring NO_3^- leaching in free-draining alluvial soils. It also emphasises the need for a high number of replications or better characterisation of urine deposition to address the practical difficulties in accurately detecting NO_3^- leaching in a functioning dairy farm. The trench lysimeters used in this study performed well in detecting drainage quantities of each treatment, however, spatial variation in urine deposition resulted in high variability in NO_3^- -N concentrations and loads between lysimeters.

The Div-Con treatment constantly measured the highest NO_3^- -N loads over two years; however, due to anomalies associated with two lysimeters under this treatment, it is difficult to conclude this treatment effect. The Div-Reg treatment measured the lowest NO_3^- -N loads over the two-year study, with loads that were measured at only 21.4% and 20.8% of the Std-Con treatment in 2023 and 2024, respectively.

Given these challenges, together with the short-term duration of the current study, we need to be cautious when determining any treatment effects at this stage. Therefore, it is crucial to perform longer-term comprehensive monitoring to accurately assess the impact of different treatments on NO_3^- -N leaching in dairy-grazed pastures, considering the various influencing factors that have been highlighted in the current study.

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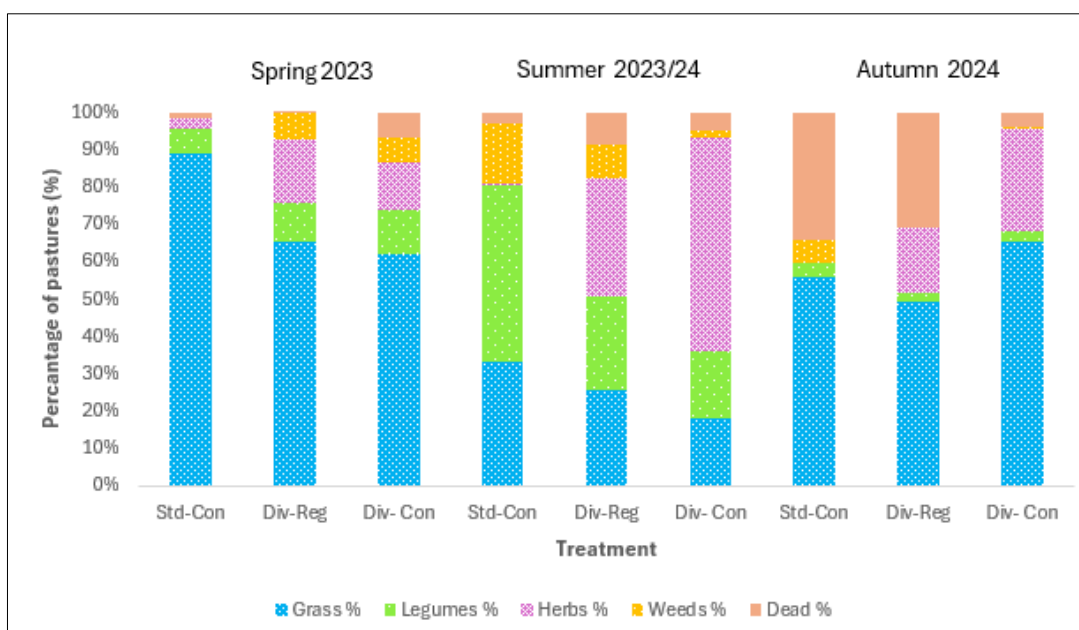
Appendices:

Appendix 1: Drainage area of each lysimeter after field establishment.

Lysimeter width (m)	Lysimeter length (m)	Drainage area (m²)
0.309	10.60	3.275
0.303	10.40	3.151
0.342	10.15	3.471
0.320	10.20	3.264
0.353	10.05	3.548
0.333	10.10	3.363
0.333	10.15	3.380
0.317	10.00	3.170
0.316	10.00	3.160
0.323	10.10	3.262
0.316	9.90	3.128
0.319	10.05	3.206

Appendix 2: Pasture botanical composition of measured from lysimeter paddocks in 2023 and 2024.

Sample Date	Paddock	Season	Grass %	Legumes %	Herbs %	Weeds %	Dead %
10/11/2023	Std-Con	Spring 2023	88.85	7.01	2.53	0.00	1.61
10/11/2023	Div-Reg	Spring 2023	65.55	10.31	16.89	7.06	0.19
10/11/2023	Div- Con	Spring 2023	56.58	12.86	17.39	13.07	0.10
30/11/2023	Div- Con	Spring 2023	67.71	10.46	8.67	0.30	12.86
8/01/2024	Div-Reg	Summer 2023/2024	36.44	35.55	18.11	1.67	8.24
1/02/2024	Std-Con	Summer 2023/2024	35.24	46.17	0.00	13.63	4.96
5/02/2024	Std-Con	Summer 2023/2024	31.33	48.20	1.10	18.56	0.81
19/02/2024	Div-Reg	Summer 2023/2024	14.66	15.00	45.48	15.58	9.28
19/02/2024	Div-Con	Summer 2023/2024	18.10	18.01	57.16	2.12	4.62
29/04/2024	Std-Con	Autumn 2024	56.03	3.88	0.00	5.96	34.12
29/04/2024	Div-Reg	Autumn 2024	49.11	2.36	17.64	0.00	30.89
29/04/2024	Div-Con	Autumn 2024	65.48	2.98	27.13	0.43	3.98



Appendix 3: Nitrate load (kg N/ha/yr) and standard error associated with each load event in 2023.

Analysis date	Std-Con		Div-Reg		Div- Con	
	Cumulative NO ₃ ⁻ -N load (kg/ha)	SE	Cumulative NO ₃ ⁻ -N load (kg/ha)	SE	Cumulative NO ₃ ⁻ -N load (kg/ha)	SE
24/02/2023	2.38	-	0.15	-	0.00	-
19/05/2023	4.12	±2.2	0.17	±0.1	0.00	±0.0
7/06/2023	7.59	±3.6	0.56	±0.5	1.99	±0.8
26/06/2023	7.77	±3.7	0.57	±0.5	2.13	±0.8
7/07/2023	7.84	±3.7	0.58	±0.5	2.18	±0.8
17/07/2023	8.60	±4.1	0.73	±0.6	2.52	±0.8
28/07/2023	8.86	±4.2	0.75	±0.6	2.82	±0.9
3/08/2023	9.10	±4.3	1.04	±0.6	3.44	±1.1
11/08/2023	9.76	±4.4	1.54	±0.8	10.43	±6.4
18/08/2023	10.16	±4.3	2.54	±1.2	19.09	±10.1
25/08/2023	12.34	±5.7	3.02	±1.5	21.46	±10.9
31/08/2023	12.46	±5.8	3.15	±1.6	22.25	±11.1
4/10/2023	13.20	±6.4	3.21	±1.7	22.79	±11.2
19/10/2023	13.49	±6.5	3.24	±1.7	23.00	±11.3
27/10/2023	13.49	±6.5	3.25	±1.7	23.24	±11.4
21/11/2023	13.67	±6.5	3.30	±1.7	23.54	±11.5

Appendix 4: Nitrate load (kg N/ha/yr) and standard error associated with each load event in 2024

Analysis date	Std-Con		Div-Reg		Div- Con	
	Cumulative NO ₃ ⁻ -N load (kg/ha)	SE	Cumulative NO ₃ ⁻ -N load (kg/ha)	SE	Cumulative NO ₃ ⁻ -N load (kg/ha)	SE
21/08/2024	6.06	-	1.09	-	8.27	-
28/08/2024	8.90	±1.8	1.77	±0.3	14.68	±6.0
2/09/2024	16.69	±5.1	3.64	±0.9	30.18	±10.1
6/09/2024	18.32	±5.8	4.04	±1.1	32.63	±11.8
11/09/2024	19.01	±6.1	4.24	±1.2	34.73	±12.6
16/09/2024	20.23	±6.9	4.46	±1.4	36.02	±13.0
18/09/2024	20.80	±7.2	4.62	±1.4	37.52	±13.8
23/09/2024	23.29	±8.6	5.13	±1.7	41.40	±15.6
16/10/2024	23.69	±8.8	5.26	±1.8	42.26	15.8
30/10/2024	23.85	±8.9	5.28	±1.8	43.22	±16.3
4/11/2024	23.96	±9.0	5.39	±1.8	43.64	±16.4
18/11/2024	24.11	±9.0	5.42	±1.8	43.93	±16.5